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Mini-Conference

ADVANCES IN WATER QUALITY MODELING

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Co-Sponsored by the
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PREFACE

There were two main objectives in holding this mini-conference. First, it provided an opportunity for ASAE members and others to showcase the results of research and educational projects relating to advances in water quality modeling. In the past decade there have been rapid advances in computer technologies and our ability to acquire data. Information has become more affordable and more readily available. Also, the Internet, GPS, GIS, remote sensing technologies, multi-media capabilities, and high powered inexpensive PC computers and workstations, has lead to the development of improved hydrologic models, the application of advanced statistical procedures, the availability of extensive data sets, and more interesting and useful methods for disseminating information.

The second objective was to build on an improved format for ASAE meetings. In recent years ASAE has undergone much reorganization. We have moved to having a single meeting and many committees and groups in ASAE have reorganized. However, the general format of the annual meeting has changed little. Often technical sessions are a forum for graduate students, scientists, and engineers to provide progress reports on their research activities. There are too many no-shows and often the presenters fail to prepare a paper for distribution at the meeting. The papers which are available are usually lengthy and often hastily prepared. Like other societies ASAE continues to sponsor and conduct Special Conferences which are held independently of the annual meeting. The quality of these conferences is usually very high but they are expensive to organize and attend. It typically takes 3-4 years to organize a specialty conference and papers are usually prepared at least six months before the conference.

This same format was used for the 1997 meeting and we felt that it was very successful. We hope you find these proceedings as valuable. We would like to express our sincere appreciation to the many people who helped organize this min-conference, and the sponsors, presenters and participants.

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SPONSORSHIP AND COOPERATING ORGANIZATIONS

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SW-21 Hydrology Group which is part of the Soil and Water Division of ASAE
USDA-ARS Grassland Soil and Water Research Laboratory

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The chair of the mini-conference and SW-21 Hydrology Group would like to acknowledge and express their sincere appreciation to the presenters, participants and many people who assisted in organizing the conference. The session agendas were developed with the help of James Converse, Soil and Water Session Organizer, David Bosch chair of the SW-21 Hydrology Group, members of the SW-21 Hydrology Group, and the many people who expressed interest in presenting a paper or poster. We would also like to thank the keynote speakers and industry participants.

Thanks are extended to Sharon McKnight and Donna Hull, at ASAE Headquarters for their help, expertise and willingness to accommodate the requests of the mini-conference chair.

Particular thanks are extended to the many people who served on the Awards and Presentation Panel. Typically, they read and judged between 3-5 papers and devoted much time to this initiative.

Special thanks are extended to Georgie Mitchell, environmental science technician for the USDA-ARS. She coordinated all aspects of the publication process. She prepared the proceedings and awards, retyped and formatted several papers, prepared cover pages for several papers and served as primary liaison with the presenters when there was a problem with the manuscripts. We could not have successfully held the mini-conference without her dedication in this effort. We would also like to thank the many other people at USDA-ARS, Grassland Soil and Water Research Laboratory that assisted in this publication.

PROCEEDINGS AND MINI-CONFERENCE FEEDBACK

We would like to hear from you. Please provide any feedback on this mini-conference to ASAE, Kevin King, or any member of SW-21 Hydrology Group. Addition copies of the proceedings can be obtained from ASAE Headquarters:

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PART 1 - ADVANCES IN WATER QUALITY MODELING I

SESSION MODERATOR:

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Coupling Mixing Zone Concept with CDE to Model Chemical Transport from Soil Solution to Surface Runoff [1]

Abstract

Modeling chemical transfer from soil to surface runoff is essential for developing a surface water quality model that can be used to assess pollution potentials of agricultural chemicals. Chemical transfer to runoff can be modeled as a two-rate process. A fast rate subprocess, which prevails at early stages of rainfall, causes an exponential depletion of chemicals from the mixing zone. A slow rate subprocess, which becomes significant under poor drainage conditions, transports chemicals into the mixing zone from the soil below. The two-rate process can be described by coupling the mixing zone concept with the convection-diffusion equation (CDE). We evaluated this coupling approach by comparing predicted results with measured bromide concentration data. A finite element scheme was developed to solve the CDE in conjunction with a near-surface boundary condition derived from the mixing concept. Overall results showed the coupling approach satisfactorily predicted bromide concentrations in both surface runoff and soil solution. The successful applications of this coupling approach indicate that the identification of the fast and slow rate subprocesses is useful. The separation of the two rate subprocesses not only allows for refinement of the mixing zone concept for use under poorly drained conditions, but also allows for direct application of the mixing concept under free drainage conditions.

Keywords: chemical runoff, surface water quality modeling, solute transport

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Introduction

Agricultural chemicals such as fertilizers and pesticides are major nonpoint source pollutants that deteriorate surface and subsurface water quality. Polluted surface runoff or water bodies such as lakes and ponds can contaminate groundwater either through direct contact with groundwater table or macropore flow. Thus, a thorough understanding of mechanisms involved in chemical transfer from soil solution to surface runoff can further the development of mathematical models that can be used to predict pollution potentials of nonpoint source pollutants and to lay out the best management plans for controlling surface and subsurface water contamination.

The conventional mixing zone concept assumes that there exists a region below the soil surface in which rainwater, soil solution, and infiltrating water mix completely and instantaneously and that there is no chemical transfer into that region from the soil below. This concept has been used to predict chemical loss in surface runoff (Donigian *et al.*, 1977; Frere *et al.*, 1980; Ahuja *et al.*, 1981; Steenhuis *et al.*, 1994). By further assuming that the mixing depth (z) and the saturated volumetric water content (θ_s) do not change with time, a mass balance equation for a mobile chemical with a constant ponding depth can be written as:

$$(z\theta_s + h_{\max})\frac{dC}{dt} = -RC \qquad [1]$$

where h_{max} is the maximum ponding depth; C is the chemical concentration in the mixing zone, runoff, and infiltrating water; t is the time; and R is a constant rainfall rate. A solution to Eq. [1] shows chemical concentration in the mixing zone decreases exponentially with time. Ahuja and Lehman (1983) and Snyder and Woolhiser (1985) reported that Eq.[1] was approximately valid under free infiltration conditions, but was invalid under poorly drained conditions.

Zhang *et al.* (1997a) identified two transfer subprocesses in their experimental study. A fast rate subprocess, which operates on a shorter time scale and is confined to the mixing zone, causes an exponential depletion of chemicals in that zone. A slow subprocess, which becomes significant on a longer time scale and is dominated by molecular diffusion and flow mechanical dispersion, simulates chemical transfer from the underlying soil to the mixing zone. The fast rate subprocess predominates under free infiltration conditions, while the slow rate subprocess becomes significant under poorly drained conditions. In their paper, they hypothesized that this two rate process could be modeled by coupling the mixing concept with the CDE.

The objective of this study was to evaluate the applicability of the coupling approach in describing the chemical transfer from soil solution to surface runoff by comparing calculated results with measured bromide data of Ahuja and Lehman (1983).

Mathematical Descriptions

Transport of a nonreactive chemical in a soil profile can be described by the CDE as:

$$D\frac{\partial^2 C}{\partial x^2} - (R-q)\frac{\partial C}{\partial x} - \theta_s\frac{\partial C}{\partial t} = 0 \qquad [2]$$

where C is the dissolved chemical concentration in soil solution; x is the depth from the soil surface; q is the runoff rate; and D is the summation of the molecular diffusion coefficient D_s and mechanical dispersion coefficient D_h as defined below:

$$D = D_s + D_h \qquad [3a]$$

$$D_h = \varepsilon(R-q) \qquad [3b]$$

in which ε is the dispersivity. The D_s in soil may be estimated by Millington and Quirk (1961) as:

$$D_s = D_w\theta_s^{4/3} \qquad [4]$$

where D_w is the molecular diffusion coefficient in pure water. Zhang *et al.* (1997a) has shown that the assumption of no chemical transfer into mixing zone from the soil below, which was used in deriving Eq. [1], was invalid under the poorly drained conditions. Therefore, Eq. [1] must be modified to allow a chemical input into the mixing zone from the underlying soil. If we define a net or total transfer flux between the mixing zone and the underlying soil as J_n, and further allow a transient build-up of a water layer (h) at the soil surface, Eq. [1] can be modified as:

$$-qC-(h+z\theta_s)\frac{\partial C}{\partial t}=J_n \qquad [5]$$

where h is the transient ponding depth and is equal to (qt) before the maximum depth (h_{max}) is reached. Based on Eqs. [2] and [5], a flux boundary condition at the interface between the mixing zone and the underlying soil (at x = z) can be written as (Zhang *et al.*, 1997a):

$$D\frac{\partial C}{\partial x}-RC=(h+z\theta_s)\frac{\partial C}{\partial t} \qquad [6]$$

where the chemical concentration in the mixing zone was assumed to be equal to that in the underlying soil at the interface (at x = z). In this study, a zero-concentration gradient for the lower boundary condition at the bottom of a soil column ($x = L_b$) could be assumed as:

$$\frac{\partial C}{\partial x}=0 \qquad t>0, x=L_b \qquad [7]$$

A constant initial concentration of C_0 throughout the soil profile or soil column was used in this study where:

$$C=C_0 \qquad t=0, 0\le x\le L_b \qquad [8]$$

Equations [2] through [8] were solved with a finite element scheme. Assuming that the runoff area is small enough to be treated as a uniform point source, solutions at $x\le z$ represent the chemical concentrations in surface runoff . To simplify the scheme, the mixing zone was treated as a super diffusion layer in which a fictitious diffusion coefficient was used to ensure a complete mixing when solving the governing equation of Eq. [2]. Thus, the storage change within the mixing zone can be dropped from Eq. [6], and the flux boundary condition at the soil surface (x = 0) can be rewritten as:

$$D\frac{\partial C}{\partial x}=RC+h\frac{\partial C}{\partial t} \qquad [9]$$

This boundary condition allows for a gradual accumulation of a well stirred water layer at the soil surface before a maximum ponding depth (h_{max}) is established. A constant value of h_{max} is used afterwards.

A finite element method was formulated to obtain numerical solutions. The element matrices were derived by evaluating the residual integral for each element by using a continuous piecewise smooth linear equation. The global matrices were obtained by incorporating element matrices using the direct stiffness method (Segerlind, 1984). The central difference method was used to approximate solute concentration for each element at different time steps. A computer program written in C^{++} was used to solve the final system of equations.

Model Application and Discussion

Experimental data of Ahuja and Lehman (1983) from two soils (Ruston fine sandy loam and Ruston loam soils (fine-loamy, siliceous Typic Paleudult)) under three infiltration conditions (zero, restricted, and free infiltration) were used to evaluate this model. In their experiments, a 61-min rain with a mean rainfall intensity of 68 mm/h was delivered to a pre-saturated soil box (100 by 15 by 10 cm), where the soils were initially equilibrated with 4000 mg/L Br solution. During each rain, surface runoff was collected for flow rate and Br concentration determination at the time of runoff initiation and periodically thereafter. At the end of each rain, the soil in the box was sampled from depth intervals of 0 to 2.5, 0 to 5, 5 to 10, 10 to 20, 20 to 30, 30 to 40, and 40 to 50 mm below the soil surface. Samples from each interval were composited and analyzed for Br concentration.

The finite element scheme, which couples the mixing zone concept with the CDE, was used to predict chemical concentrations in both runoff and soil solution for the zero and restricted infiltration conditions. An analytical solution to Eq. [1], which is solely based on the mixing zone concept, was used under the free infiltration conditions. A constant effective mixing zone depth of 2 mm and an average maximum ponding depth of 0.5 mm were used in the calculation for both soils. The former was based on the previous work of Ahuja *et al.* (1981), who found the effective mixing depths for Ruston fine sandy loam and other two soils were between 2 and 3 mm. We chose 2 mm based on our experience and experimental observations. The average maximum ponding depth at steady state was computed as a mean flow depth weighted by flow length. A fictitious unit value of D was used within the mixing zone to ensure a complete mixing in that zone. The other parameter values used in the model prediction are given in table 1. Except for D_h, which was estimated by comparing measured with predicted data in this study, the rest of the data were either

measured or computed independently by Ahuja (1990).

Predicted chemical concentrations in surface runoff under three infiltration conditions for Ruston fine sandy loam using the proposed coupling model and the mixing model of Eq. [1] were compared with those of measured in figure1.

Table 1. Measured and estimated parameter values used in the numerical solutions.

Soil	Infiltration Treatment	θ_s cm^3/cm^3	C_0 mg/L	R cm/s	q cm/s	D_s cm^2/s	D_h cm^2/s
Ruston fine sandy loam	Zero	0.53	4000	1.98×10^{-3}	1.98×10^{-3}	5.15×10^{-6}	0
	Restricted			1.97×10^{-3}	1.89×10^{-3}	5.15×10^{-6}	9.85×10^{-6}
	Free			1.79×10^{-3}	1.03×10^{-3}	NA	NA
Ruston loam	Zero	0.49	4000	2.00×10^{-3}	2.00×10^{-3}	4.64×10^{-6}	0
	Restricted			1.96×10^{-3}	1.91×10^{-3}	4.65×10^{-6}	5.36×10^{-6}
	Free			2.02×10^{-3}	1.63×10^{-3}	NA	NA

In general, the predictability of the simple mixing model increased progressively as infiltration rates increased from impervious to free drainage conditions. A close agreement between predicted and measured data under free infiltration conditions (figure 1c) indicated that Eq. [1], which assumes there is no chemical transfer into the mixing zone from the underlying soil layer, was valid under these conditions, and it further indicated that there was no need to use complicated models to predict chemical concentrations in surface runoff under these conditions. Similar conclusions have been drawn by Ahuja and Lehman (1983) and Ahuja *et al.* (1981). Zhang *et al.* (1997b) also reported that Eq. [1] produced satisfactory predictions for moderately adsorbed herbicides if a desorption /adsorption isotherm was included. However, the mixing model considerably underpredicted Br concentrations under zero and restricted infiltration conditions (figures 1a and 1b). This is because the upward chemical transfer into the mixing zone by molecular diffusion and mechanical dispersion cannot be ignored under poorly drained conditions.

Coupling the CDE with the mixing model provided a better solution under zero and restricted infiltration conditions. Under zero infiltration conditions where the molecular diffusion coefficient (D_s) was used, numerical solution predicted well at early stages of runoff. However, it underpredicted chemical concentrations at large time scales. The underprediction could be caused by several reasons. First, chemicals retained inside the soil aggregates could not mix completely and instantaneously with soil solution as assumed in the mixing zone, and might be slowly released into soil solution by the soil aggregates through molecular diffusion. Secondly, it was possible that chemicals could be transported to the soil surface by subsurface lateral flow at 4% slope under zero infiltration conditions (Zhang *et al.*, 1997a).

We didn't calibrate the proposed coupling model by adjusting the diffusion coefficient D of Eq. [2] to match measured data for zero infiltration conditions. The input parameter values used in the prediction were either directly measured or independently computed using Eq. [4] (table 1). Therefore, the better prediction over the mixing model indicated the improved model predictability, rather than the feasibility of fitting the data with this model. Another reason for not adjusting D was because the contributions from subsurface lateral flow or release by soil aggregates were unknown. Under restricted infiltration conditions, since the mechanical dispersion caused by downward movement of infiltrating water could be significant as suggested by Wallach and van Genuchten (1990), the D_h of Eq.[2] was calibrated to match the experimental data at large time scales (figure 1b). The measured data were satisfactorily reproduced by the proposed model using the calibrated D_h (table 1). The dispersivity (ε), estimated by dividing D_h by infiltration flux in our notations, was about 1.6 cm, which was within ranges for disturbed or repacked soil columns (van Genuchten and Wierenga, 1986).

Measured and predicted chemical concentrations in the soil solution at the end of 60 min rainfall are plotted with soil depth in figure 2. For both zero and restricted infiltration conditions, the predicted concentrations agreed qualitatively well with measured values. As suggested by Ahuja (1990), more stringent quantitative comparison was not suitable due to remarkable sampling errors expected due to the nature of the sampling procedures. No plots are presented for free infiltration conditions, since the simplified mixing zone approach of Eq. [1] is unable to predict concentration distribution with soil depth.

Measured and predicted chemical concentrations for Ruston loam soil are not shown. The general trends of model predictability were similar to those of Ruston fine sandy loam under all three infiltration conditions. That is, the

predictability of the simple mixing model increased progressively as infiltration rates increased from impervious to free drainage conditions. Comparatively, a closer agreement between predicted and measured data at large time scales was obtained for zero infiltration conditions with the molecular diffusion coefficient (D_s) given in table 1. Again, the addition of a calibrated D_h caused a rightward shift in the early prediction curve for restricted infiltration. This indicates

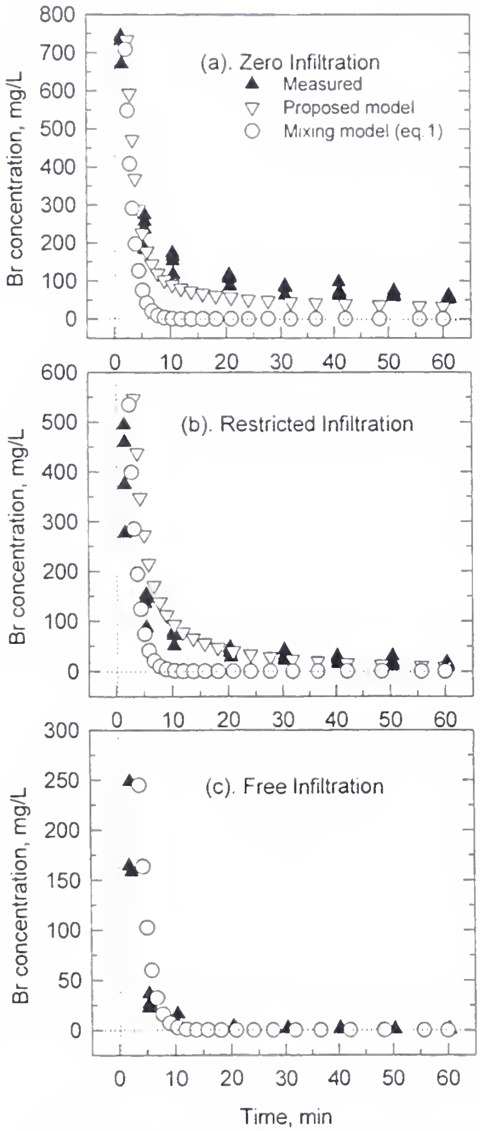

Figure 1. Measured and predicted Br concentrations in surface runoff under three infiltration conditions for the Ruston fine sandy loam soil.

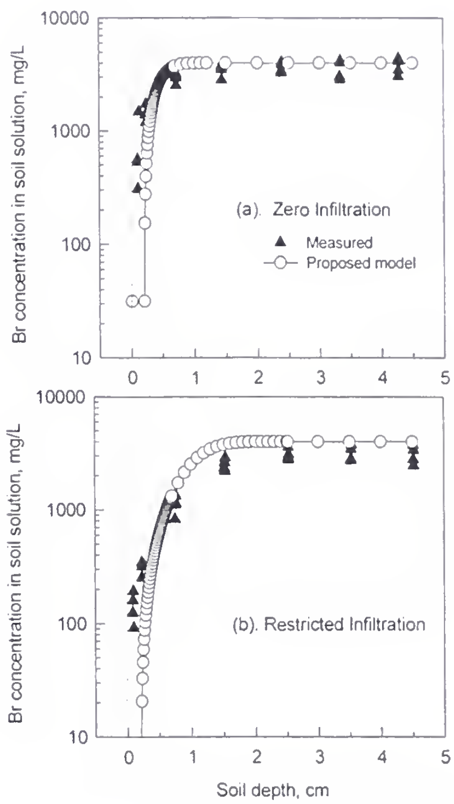

Figure 2. Measured and predicted Br concentration profiles in soil solution at the end of 60-min rains for the Ruston fine snady loam.

that in addition to the mechanical dispersion transport, other chemical sources might also contribute to the soil solution of the mixing zone. The possible sources could be the release of chemicals by soil aggregates or/and transport of chemicals to the mixing zone by lateral flow. The spatial distributions of chemical concentrations in soil solution with depth at the end of rainfall for Ruston loam were not shown. The general agreements between measured and predicted data were similar to those of Ruston fine sandy loam, indicating that the model predicted concentration profiles reasonably well for both soils.

Conclusions

This study demonstrates that nonreactive chemical transfer from soil solution to surface runoff can be described as a two-rate process. A fast rate subprocess, which is limited to the mixing zone (<2-3 mm), is responsible for an exponential depletion of chemicals in the zone. A slow rate subprocess, which can be simulated by the conventional CDE, describes chemical transfer into the mixing zone from the soil below. The fast rate subprocess prevails at earlier stages of rainfall, while the slow subprocess becomes predominant at later stages, especially for the cases when a perched water table is close to the soil surface (Zhang *et al.*, 1997a). Successful applications of the coupling approach, which combines the mixing zone concept with the CDE, indicate that the identification of the fast and slow rate subprocesses is useful. The separation of the two rate subprocesses not only allows for refinement of the mixing zone concept for use under poorly drained conditions, but also allows for direct application of the mixing concept under free drainage conditions.

The overall results showed that the coupling approach satisfactorily predicted chemical concentrations in both surface runoff and soil solution for nonreactive chemicals under poorly drained conditions. However, further tests on reactive chemicals such as herbicides are needed. The finite element scheme used in this study can be easily modified for this purpose by incorporating an adsorption/desorption isotherm into the formulation.

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Model of the Transport of Water and Nutrients Through the Vadose Zone[1]

Abstract

Many regulatory agencies are striving to develop water management programs to minimize the accumulation of agricultural chemicals in groundwater and to implement conservation measures to optimize use of groundwater resources. These programs often use models to predict the consequences of alternatives. However, most modeling exercises do not explicitly link actions on the soil surface to groundwater impacts. The transport of water and crop nutrients through the vadose zone is also neglected in most studies. The dynamics of aquifer response to water management is inaccurate if the vadose zone is neglected. The anticipated change in the groundwater systems may be correct. However, due to the delay for transport in the vadose zone the modeled outcomes are forecasted to occur too quickly. This leads to policies that cannot fulfill the modeled response in the time frame originally predicted. We developed an analytical solution that can be combined with a groundwater model to provides for the delay in transport through the vadose zone.

We used the method of decomposition to develop an analytical solution to the Richards equation for water flow in unsaturated soils. The method is relatively simple to use. The procedure allows for repetitive use to integrate the variability of deep percolation across the soil surface and to allow for integration for the cells used to model groundwater. The current formulation requires iteration to select the surface boundary condition for the vadose zone so that the infiltration into the vadose zone matches the deep percolation from crop root zones. We have also included a solute transport routine, based on transfer functions, to predict the arrival of crop nutrients to the groundwater.

The vadose zone model was compared to the HYDRUS numerical model for a range of conditions. Results showed that the pattern of flow past a point in the vadose zone was similar as long as the correct volume of inflow was used for the vadose zone model. While the vadose zone model is more limited than simulation models like HYDRUS, it is also simpler to use. In that sense, the vadose zone model fulfills the objective for which it was developed.

We are continuing research to develop methods to improve the analytical solutions. We are confident that this will improve modeling of management strategies for groundwater protection and conservation.

Keywords: groundwater recharge, groundwater quality

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[1] ASAE Paper 99-2065. D.L. Martin, Professor, Dep. of Biological Systems Engineering, Univ. of Nebraska - Lincoln, Lincoln, NE 68583-0726; J.L. Arumi, Assistant Profesor, Dep. of Agricultural Engineering, University of Concepcion, Chillan, Chile; D.G. Watts, Professor, Dep. of Biological Systems Engineering, Univ. of Nebraska - Lincoln, Lincoln, NE 68583-0726. Contact author (dmartin2@unl.edu).

Introduction

The vadose zone retards the transport of water and crop nutrients from the crop root zone to groundwater. Transport may require months, or even years, depending on the depth and properties of the soils in the vadose zone. During transit the vadose zone acts as a reservoir, buffering the root zone drainage resulting from hydrologic and irrigation events, yet many groundwater quality studies ignore these effects. Consequently, many programs understate the difficulty and expense of minimizing the impacts of agricultural practices on groundwater quality and quantity. Our objective was to develop a simplified method to estimate the travel time for water and nitrate-nitrogen in the vadose zone which we are defining as the unsaturated soil beneath the root zone. The procedure provides a linkage between a watershed model that integrates field practices and a groundwater model used to simulate the effect of agricultural management on groundwater quantity and quality.

Several considerations pertain when simulating transport in the vadose zone. The difference between the domain of the watershed-plant model (WPM) and the groundwater model is important. The groundwater model consists of a finite difference program with a cell size of 800 m x 800 m. In contrast, elements in the watershed-plant model are homogeneous crop and soil units that are managed similarly. The recharge, calculated by the WPM, must be spatially integrated for each groundwater model cell. Additionally, the WPM uses a daily time step compared to monthly for the groundwater model. Spatial and temporal integration requires multiple simulations of the vadose zone for each groundwater cell; thus, the VZM must be efficient. The vadose zone model (VZM) must estimate the cumulative volume of water that reaches a given depth in the vadose zone, after a known volume of water has drained from the root zone. In this way, the output of WPM can be used to estimate groundwater recharge.

Model Development

Since, deep percolation of water is predominantly a one-dimensional process, we focused on solution of the Richards equation for a vertical column. When written using diffusivity the Richards equation becomes:

$$\frac{\partial\theta}{\partial t}=\frac{\partial}{\partial z}\left[D(\theta)\cdot\frac{\partial\theta}{\partial z}\right]-\frac{\partial K(\theta)}{\partial\theta}\frac{\partial\theta}{\partial z} \qquad [1]$$

where z is the elevation (positive downward), t is time, $\theta(z, t)$ is the volumetric soil-water content, $D(\theta)$ is the soil water diffusivity (hydraulic conductivity/soil water storage capacity), and $K(\theta)$ is the vertical hydraulic conductivity.

Expressions for soil water retention and unsaturated hydraulic conductivity are needed to solve Richards equation. The mathematical expressions developed to represent those processes are highly non-linear. We considered the Brooks and Corey (1966) and the van Genuchten (1980) water-retention and hydraulic-conductivity functions. These are widely used for unsaturated flow. In addition, parameters for the functions can be estimated using physical properties. In general, the van Genuchten (VG) model matches experimental data over a wider range of water contents than the Brooks and Corey (BC) functions. Also, many researchers present data in terms of VG parameters. However, the VG functional is complicated limiting its use for analytical solutions, while the BC expressions are easier to manipulate.

Even though the Richards equation is highly non-linear and analytical solutions are not easy to obtain, there have been several recent advances. Broadbridge and White (1988) presented an analytical solution for a semi-infinite soil column, with a uniform initial soil water content and a constant recharge. Warrick et al. (1990) presented an extension of the solution of Broadbridge and White that did not require a constant initial water content. Warrick et al. (1991) presented a second extension that permits the evaluation of time variable recharge. The solutions by Broadbridge and White (1988), and extended by Warrick et al. (1990 and 1991) use a dimensionless form of the Richards equation and involve a series of mathematical transformations. This methodology is difficult to use and the system of equations can diverge as the water profile nears a unit gradient.

Serrano (1998) used the method of decomposition to develop an approximate solution to Richards equation. Serrano (1998) assumed that the diffusivity approaches a maximum value near saturation; however, the specific moisture capacity approaches zero near saturation and the diffusivity becomes infinite. We assumed that the diffusivity reaches a maximum value (Ds) near the bubbling pressure where the soil water content approaches saturation and the hydraulic conductivity approaches the saturated value. These simplifications lead to the following conditions:

$$D(\theta\rightarrow\theta s)\approx Ds \qquad \text{and} \qquad \frac{\partial K(\theta)}{\partial\theta}=0 \qquad [2]$$

These relationships give a linear approximation for the Richards equation:

$$\frac{\partial\theta}{\partial t} = Ds\cdot\frac{\partial^2\theta}{\partial z^2} \qquad [3]$$

subject to the following boundary conditions:

i) The top boundary condition is a series of pulses of duration Tj using the Heaviside step function Φ:

$$\theta(0,t) = \sum_j \Phi(t - T_j)\cdot\Delta\theta_j \qquad [4]$$

$$\partial\theta(z,t)/\partial z \rightarrow 0 \quad \text{as } z\rightarrow\infty$$

ii) Semi-infinite soil column with a unit gradient at the lower boundary

iii) Constant initial volumetric soil water content profile; $\theta(z,0)=\theta i$

The analytical solution for the linear approximation is:

$$\theta_0(z,t)=\sum_j\Delta\theta_j\cdot\mathrm{erfc}\left(\frac{z}{\sqrt{4\cdot Ds\cdot(t-T_j)}}\right)\cdot\Phi(t-T_j)+\theta_i \qquad [5]$$

The solution is for the linear component of the Richards equation. The non-linear components are close to zero near saturation but become more important as the soil dries (Serrano, 1998). An auxiliary function was developed for the non-linear components. The difference between Ds and the diffusivity for drier conditions is given by:

$$Dp(\theta) = Ds - D(\theta) \qquad [6]$$

Substituting these relationships into Richards equation and rearranging gives:

$$\frac{\partial\theta}{\partial t} - Ds\cdot\frac{\partial^2\theta}{\partial z^2} + \frac{\partial Dp}{\partial\theta}\cdot\left(\frac{\partial\theta}{\partial z}\right)^2 + Dp\cdot\frac{\partial^2\theta}{\partial z^2} + \frac{\partial K}{\partial\theta}\cdot\frac{\partial\theta}{\partial z} = 0 \qquad [7]$$

In a fashion similar to Serrano (1998), the solution for the Richards equation can be approximated as:

$$\theta=\theta o-\int_0^t \sum_{n=0}^{\infty} An\, dt \qquad [8]$$

where θo is the solution determined for the linear portion in equation 5. The series is defined as:

$$Ao = N(\theta o) \qquad [9]$$

$$A1 = \theta 1\cdot\frac{d}{d\theta o}N(\theta o) \qquad [10]$$

$$A2 = \theta 2 \cdot \frac{d}{d\theta o} N(\theta o) + \frac{\theta 1^2}{2} \cdot \frac{d^2}{d\theta o^2} N(\theta o) \qquad [11]$$

and, $N(\theta)$ incorporates the non-linear terms of the solution:

$$N(\theta) = \frac{\partial Dp}{\partial \theta} \cdot \left(\frac{\partial \theta}{\partial z}\right)^2 + Dp \cdot \frac{\partial^2 \theta}{\partial z^2} - \frac{\partial K}{\partial \theta} \cdot \frac{\partial \theta}{\partial z} \qquad [12]$$

The series coverages rapidly and A2 is almost negligible. It is necessary to incorporate BC functions for the relationship between hydraulic conductivity, dispersion, and soil-water content. Numerical integration is used to evaluate the series.

With the VZM it is possible to predict the volumetric water content as a function of depth and time. However, to estimate recharge and residence time, it is necessary to compute the volume of water that reaches a depth at a specific time. The Darcy velocity of flow (v) was computed from a finite difference approximation of the hydraulic gradient. The cumulative volume of water, V(t), that passes a point z_o is then given by:

$$V(t) = \int_0^t v \, dt \qquad [13]$$

To simulate the transport of nitrate a Fickian transfer function procedure was used. The solution used the mean water flow velocity as determined from the VZM. Simulated break through curves for nitrate transport with the VZM were similar to simulated transport with a numerical model.

Results

To evaluate the VZM we compared the output from the VZM to that from the HYDRUS-2D numerical simulation model (Simunek et al., 1996). HYDRUS-2D includes the SWMS_2D model, a WINDOWS' user interface, and an automatic finite-element mesh generation program (MESHGEN). SWMS_2D is a finite element solution of the Richards equation for saturated and/or unsaturated flow. It was necessary to convert the van Genuchten parameter values used by HYDRUS to equivalent values for the Brooks and Corey functions used in the VZM. Other researchers have converted parameters by equating the specific moisture capacities.

The HYDRUS and VZM models were used to simulate water transport for a series of pulses at the top boundary of the soil column. Pulses at the top of the vadose zone allow the VZM to simulate monthly percolation from the root zone. Observation points were located at the 2, 5 and 10 m depths. We simulated three soils: sand, loam and silt. We used the hydraulic conductivity and diffusivity values suggested in the database of HYDRUS (Table 1) for each soil.

Table 1. Values of the van Genuchten parameters used for the simulations.

van Genuchten parameter	Sand	Loam	Silt
Hydraulic conductivity [cm/hr]	29.70	1.04	0.25
Saturated soil water content [cm^3/cm^3]	0.430	0.430	0.460
Residual soil water content [cm^3/cm^3]	0.045	0.078	0.034
Alpha [cm^{-1}]	0.145	0.036	0.013
n	2.268	1.560	1.370

The pulses at the top boundary were based on simulations using the EPIC model (Williams et al., 1984). Six series of root-zone percolation were estimated. Each series corresponds to a management index for one soil. The index was created using weighted combinations of crops and irrigation practices for the Central Platte Valley of Nebraska. We used the average monthly percolation as the infiltration into the vadose zone for a three-year period. Initially we selected an upper soil water content that when combined with a unit gradient at the top of the vadose zone would infiltrate the flux of water determined from the EPIC model. This soil water content was used for the top boundary condition for both HYDRUS and the VZM models. The initial soil water profile was determined from simulation with the HYDRUS

model for a 16-year period.

Simulation results showed that the VZM model regularly infiltrated a smaller volume of water than HYDRUS (table 2). Variations in infiltration are partially due to the different soil water retention and hydraulic conductivity functions. Even though the equivalent Brooks and Corey parameters were determined from the van Genuchten relationship, the functions produce different values of hydraulic conductivity and diffusivity for a specific water content. Stankovich and Lockington (1995) evaluated the effect of the van Genuchten and Brooks and Corey functions (with equivalent parameters) on the solution of Richards equation and found that the Brooks and Corey functions predicted less infiltration. Infiltration rates are also smaller because the linearization neglects gravitational effects. The solution to the linear function (θo) only depends on the diffusivity and for longer times the infiltration rate into the vadose zone would be less than if gravitational effects were included.

The rate of infiltration into the vadose zone for a range of values for the surface boundary condition and the initial soil water content are given in figure 1. The infiltration rate for a unit gradient based on the water content at the upper boundary is also shown in figure 1. These results show that the initial soil water content is relatively unimportant; however, the surface boundary condition has a significant role. The results also show that the maximum rate that can be accomplished with the current formulation is well below the flux for a unit gradient at the soil surface.

Table 2. Simulated infiltration (mm) for HYDRUS and VZM for three soils and three years.

	Infiltrated Depth - Year 1		Infiltrated Depth -Year 2		Infiltrated Depth -Year 3	
Soil	HYDRUS	VZM	HYDRUS	VZM	HYDRUS	VZM
Sand	288	192	-27	-23	286	112
Loam	236	218	7	61	266	24
Silt Loam	246	122	58	-01	259	176

The amount of water that infiltrates into the vadose zone is known from simulations with the WPM model. Thus, the results in figure 1 were used to select the boundary condition for the VZM to infiltrate the desired water during a pulse. The total depth of water that flows past a point 5 m below the surface of the vadose zone is shown in figure 2. The unadjusted VZM model is when the same soil water content is used for the top boundary condition as for the HYDRUS model. The adjusted VZM model is for when the surface boundary condition is adjusted to provide the

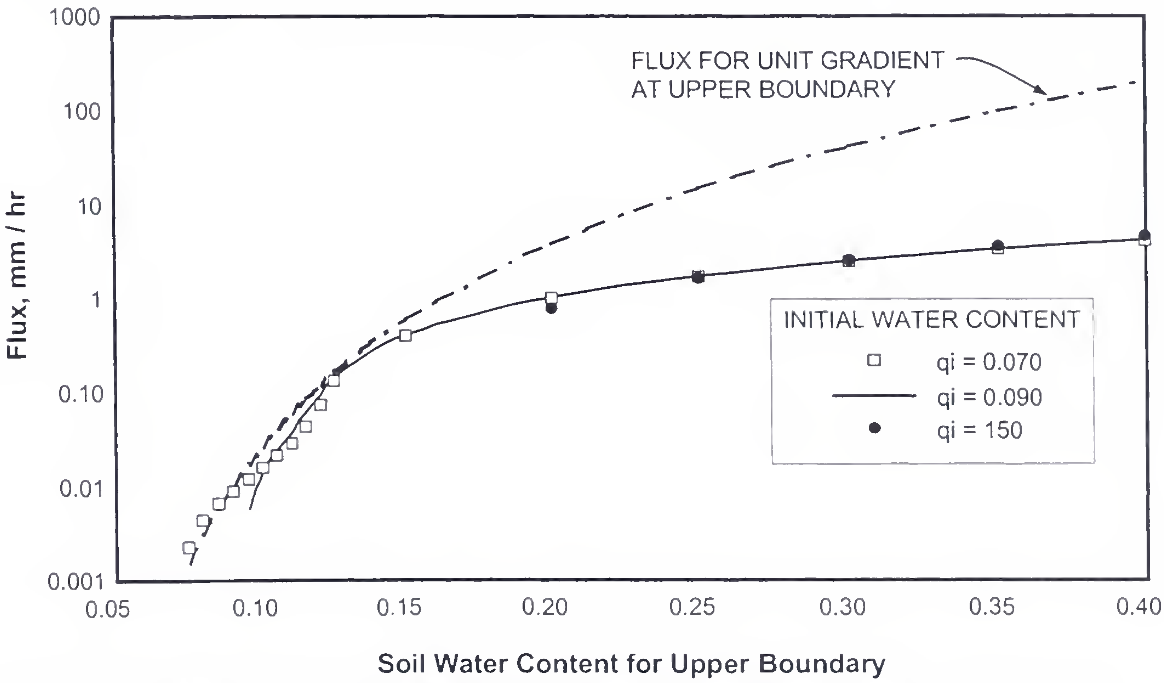

Figure 1.Infiltration fluxes from the vadose zone model for three initial soil water contents and a range of surface boundary conditions.

required depth of infiltration. Once the correct volume of water enters the soil profile, the VZM does a reasonably good job of transporting the water. It is clear that the surface boundary condition cannot be the same as for HYDRUS.

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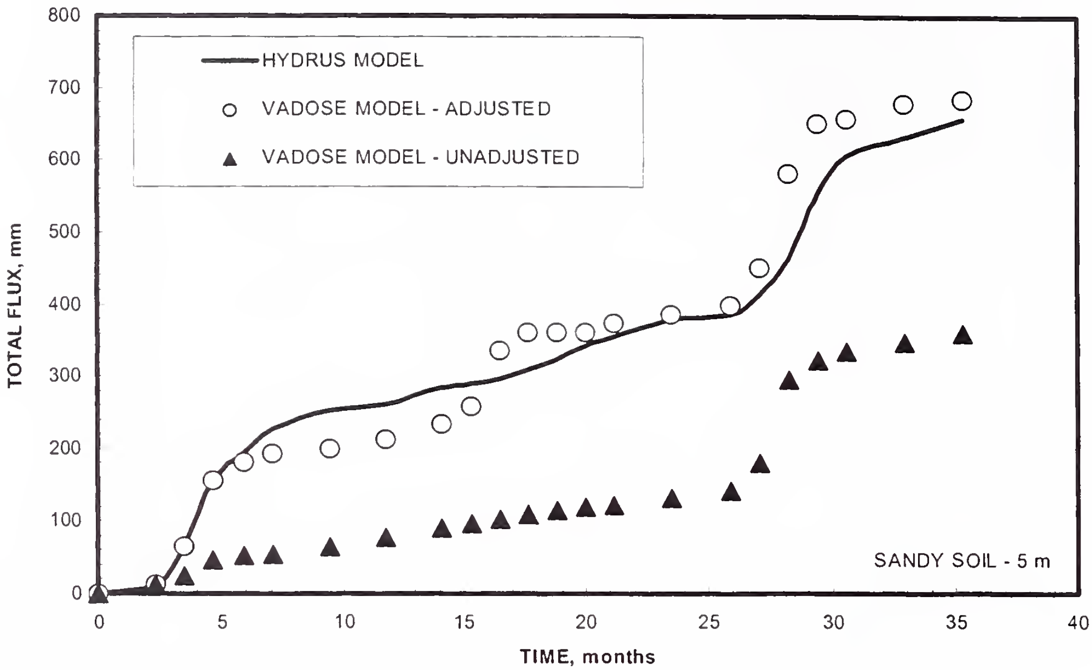

Figure 2. Simulated cumulative percolation past a 5 m depth for sand during three years.

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Prediction of Pesticide Transport Through the Vadose Zone Using Stochastic Modeling [1]

Abstract

Prediction of pesticide concentrations through the vadose zone has not been without difficulty. Existing prediction models seem to produce reasonable results for the upper portion of the root zone, but fail in accuracy at lower depths. Additionally, measurement techniques are expensive, especially when the sampling of deeper depths is desired. This study has developed a stochastic approach whereby either measured or predicted concentrations of a given pesticide in the root zone area may be used to predict the concentrations of that compound at deeper depths in the soil profile. Data collected at six depths (30, 60, 90, 120, 150, and 180 cm) were used to develop stochastic properties, such as a log normal probability density function (PDF), mean, and variance for each respective depth. Then, a relationship between mean and variances obtained for each of the six depths was developed using the method of moments. Finally, using the PDF for the 30 cm depth and adjusted mean and variances for the deeper depths, pesticide concentrations were predicted at lower depths for a completely different time period. Results indicate a great promise for accuracy and feasibility of using such a stochastic approach in predicting pesticide transport through the vadose zone.

Keywords: pesticide transport, stochastic models, solution samples

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Introduction

Characterizing pesticide movement through the vadose zone has not been without difficulty. Existing predictive models seem to produce reasonable results for upper portion of the root zone area, but fail in accuracy at lower depths. (Vyravipillai, 1994) The inherent variability of field conditions, though, causes the predictions of the models to be inaccurate (Mills and Leonard, 1984). Additionally, field measurement of parameters for input into these models is time consuming. "To truly reflect the variability inherent in potential pesticide pollution predictions it is necessary to obtain probability distributions for pollution of water resources by pesticides" (Mills and Leonard, 1984). Additionally, measurement techniques are expensive, especially when the sampling of deeper depths is desired.

A transfer function model (TFM) was proposed (Jury, 1982) to simulate solute transport under natural field conditions where substantial variability exists in water transport properties. To use the model one must first measure the distribution of travel times of a solute from the soil surface to a reference depth. The resulting distribution function is then used to simulate the average solute concentration at any given depth and time for arbitrary solute inputs or water application variability. Jury considers the soil system "in terms of its method of transforming an input function (solutes added to the soil surface) into an output function (solutes moving through the soil)". The movement of the solute is considered as a stochastic function of time or of the net amount of water applied.

The TFM model has been successfully applied to data obtained in laboratory column experiments, field studies using suction lysimeters and tile-drained field studies (Heng and White, 1995). The field-scale research of Heng and White (1996) determined that a simplified analytical form of the TFM, the TFM satisfactorily predicted the movement of sulphate through three consecutive drainage seasons in a Tokomaru silt loam (Typic Fragiaqualf) soil. Zhang (1995) examined the results of chloride transport, further demonstrating that the TFM can provide more accurate predictions than the CDE.

This study has developed a stochastic approach whereby either measured or predicted concentrations of a given pesticide in the root zone area may be used to predict the concentrations of that compound at deeper depths in the soil profile.

Materials and Methods

Study Site

The data was collected from four fields (each with an area of approximately 0.2 ha) at the Central Maryland Research and Education Center, Upper Marlboro Facility, Upper Marlboro, Maryland. The site is located in the Coastal Plain physiographic region and dominant soil at the site belongs to Monmoth fine sandy loam series with moderate infiltration characteristics (USDA Soil Survey for Prince Georges County, Maryland, 1967; new SCS classification identifies it as Marlton sandy loam, Vyravipillai, 1994). Mean saturated hydraulic conductivities (k_s) measured from samples obtained from the study site indicate great variability, with k_s values ranging from 7.9 cm/h at the surface horizon to 0.04 cm/h at 45 cm below the soil surface. The clay content of the soil increases with depth, causing decreased saturated hydraulic conductivity at lower depths.

The study site received precipitation amounts of 110.9 cm, 112.0 cm, and 87.2 cm during the 1992, 1993, and 1994 growing seasons, respectively. The average annual precipitation in the region is approximately 100 cm.

Cultural Activities

Fields 1 and 2 have been in no-till cultivation and fields 3 and 4 have been in conventional till cultivation since 1989. The fields were planted to grain corn followed by rye as a winter cover crop. The winter rye was plowed in for the conventional tilled fields whereas paraquat was used to kill the winter rye in the no till fields. Disk plows were used only on the conventionally tilled fields.

Herbicides, atrazine and alachlor, were applied to the fields in either of two forms, a liquid, wettable powder formulation (conventional formulation) or a starch encapsulated formulation. Fields 1 and 3 received the wettable powder formulation and fields 2 and 4 received the starch encapsulated formulation. Each field received active ingredients of herbicide at a rate of 1.7 kg/ha for atrazine and 2.8 kg/ha for alachlor. This report considers only the results related to atrazine and results are presented only for Field #2.

Data Collection and Analysis

Soil water samples were collected from suction lysimeters to detect the presence of atrazine and alachlor at depths of 30 cm through 180 cm in 30 cm increments at six locations in each of the four fields. Suction of between 0.7 and 0.8 bars was placed on the lysimeters 2 to 4 days after a rainfall of greater than 1.5 cm occurred. Soil water samples were collected approximately 24 hours after suction had been applied. Approximately 12 sets of samples were collected per year. The samples were analyzed by the USDA-ARS Hydrology Lab in Beltsville, Maryland. A field averaged atrazine concentration was calculated for each of the six depths, in each of the four fields, for each sampling date. These field average concentrations were then used for the stochastic pesticide predictions.

Theory

Jury(1982) developed the TFM to model complex soil systems in a simple manner by characterizing the solute flux exiting the soil system in terms of the solute input flux. The solute is considered to flow through a volume of soil known as the transport volume. The method considers vertical flux, chemical and biological degradation, but not the lateral flux. The general form of the TFM (Jury, 1982; Jury et al, 1986) is:

$$Q_{out}(t) = \int_0^t g(t-t')Q_{in}(t')dt' \qquad [1]$$

or

$$Q_{out}(t) = \int_0^t g(\tau)Q_{in}(t-\tau)d\tau \qquad [2]$$

where Q_{in} is the solute mass flux rate into the transport volume, Q_{out} is the solute mass flux rate out of the transport volume, t' is the entry time, t is the exit time, $\tau = t - t'$ is the solute lifetime in the volume, and $g(\tau)$ is the lifetime density function, representing the net effect of soil processes and the mode of solute input in the soil.

Jury(1982) also formulated the TFM in terms of net applied water (I) to predict concentration at a depth, L as:

$$C_L(I) = \int_0^\infty C_{IN}(I-I')f_L(I')dI' \qquad [3]$$

where the quantity inside the integral is the probability, $f_L(I')$, of the solute reaching a depth Z=L between I' and I' + dI', multiplied by the input concentration, $C_{IN}(I - I')$, of solution discharging at I'.

In our study, we set out to use the measured pesticide concentrations at the 30 cm depth to predict concentrations at lower depths, making several assumptions based on the TFM work by Jury. If successful, it is also envisioned that a transport model, such as GLEAMS, shown to provide accurate predictions at shallow depths (Vyravipillai, 1994), could be used to provide input to a stochastic model to provide predictions at lower depths.

The method of moments was used to fit the lognormal probability density function (pdf) to the data. Each distribution was defined by:

$$f_L(I) = \exp\frac{[-(\ln I-\mu)^2/\sigma^2]}{\sqrt{2\Pi}\,\sigma}*M_0 \qquad [4]$$

where, I is the cumulative precipitation (cm), μ is the mean of the distribution, σ^2 is the variance of the distribution and M_0 is the solute mass per unit area (first moment) of the distribution.

The piecewise linear distribution formed by connecting the field averaged data for each depth in each field was used to determine the solute mass per unit area, M_0, the mean depth of the solute pulse, M_1 (1st moment), and the moment of inertia (variance) about the concentration axis passing through I=0 cm, M_2 (2nd moment). This was accomplished by finding the moments of the distribution using standard equations. Next, the parameters μ and σ^2 of the lognormal pdf were calculated using the following equations:

$$\mu = \ln(T1), \text{ where } T1 = M_1 \qquad [5]$$

$$\sigma^2 = \ln(T2/T1^2), \text{ where } T2 = M_2/M_0 \qquad [6]$$

The lognormal distribution was fitted to the average atrazine concentration data for the 1992 and 1993 growing seasons. This resulted in a pdf for each growing season, for each depth, and for each field. The parameters of the lognormal distributions (μ, σ^2, and M_0) for each depth were then averaged together for each field. This created a set of field-specific pdf's representing two growing seasons of data. These pdf's were then used for prediction of pesticide concentrations at lower depths based upon the measured concentrations at the 30 cm depth for 1994.

To this effect, a set of pdf's were developed for the 30 cm measured data for the 1994 growing season for making predictions based on the 1992-1993 concentration data. The measured concentrations at lower depths for the 1994 growing season were also used to develop pdf's for comparison to the predicted concentrations.

Jury (1982) calculated the concentrations at lower depths using the transfer function equation [3]. To make our predictions, we developed a set of parameters relating the parameters of the 1992-1993 field-specific pdf's at the 30 cm depth to the values of these parameters at the lower depths in each of the fields. First, we fitted a polynomial to the μ, σ^2, and M_0 , parameters for each field to relate the measured concentrations at the 30 cm depth to the measured concentrations at lower depths. Each polynomial was then normalized. Next, another set of parameters, α, β, and η, was developed to represent the relation of the averaged measured data at the 30 cm depth to the fit polynomial. The relation of the mean, μ, was described by α, the variance, σ^2, was described by β and the relation of the first moment, M_0, was described by η. For example, the α parameter for a given depth was developed by dividing the value of the fit μ parameter at that depth by the fitted lognormal pdf parameter value at the 30 cm calibration depth.

$$\alpha_z = \mu_{z\,\text{fit}} / \mu_{30\,\text{fit}} \qquad [7]$$

The parameters β and η were defined in a similar manner as:

$$\beta_z = \sigma^2_{z\,\text{fit}} / \sigma^2_{30\,\text{fit}} \qquad [8]$$

$$\eta_z = M_{0z\,\text{fit}} / M_{030\,\text{fit}} \qquad [9]$$

Finally, the parameters of the lognormal pdf fitted to the 1994 measured concentrations for the field averaged measured concentrations at the 30 cm depths for the four fields were multiplied by the α, β, and η parameters to yield parameters of the pdf's for the lower depths for the 1994 growing season. For example, to predict the μ of the lognormal distribution for the 60 cm depth in 1994, the measured value of μ was multiplied by the α parameter as follows:

$$\mu_{z\,\text{pred}} = \mu_{30\,\text{calib}} * \alpha_z \qquad [10]$$

These predicted concentrations for 1994 were then compared graphically to the pdf's fitted to measured concentrations for the lower depths for the 1994 growing season.

Results

Results for Field #2 (no-till, starch encapsulated formulation) are plotted in Figure 1 for each of the six depths. The averaged measured concentration data for 1994 is plotted with a bar indicating ± one standard deviation. Graphs representing depths where predictions were made (60 cm to 180 cm) have a dashed curve representing the predicted pdf developed by multiplying the 1994 measured concentration data by the α, β, and η parameters. Each graph has a solid curve representing the lognormal pdf fitted to the 1994 measured data. The predictions are compared to the measured data graphically at this time, analyzing the shape and size of the measured and predicted pdf's. The location of the predicted pdf's in comparison to the points representing the measured data, along with

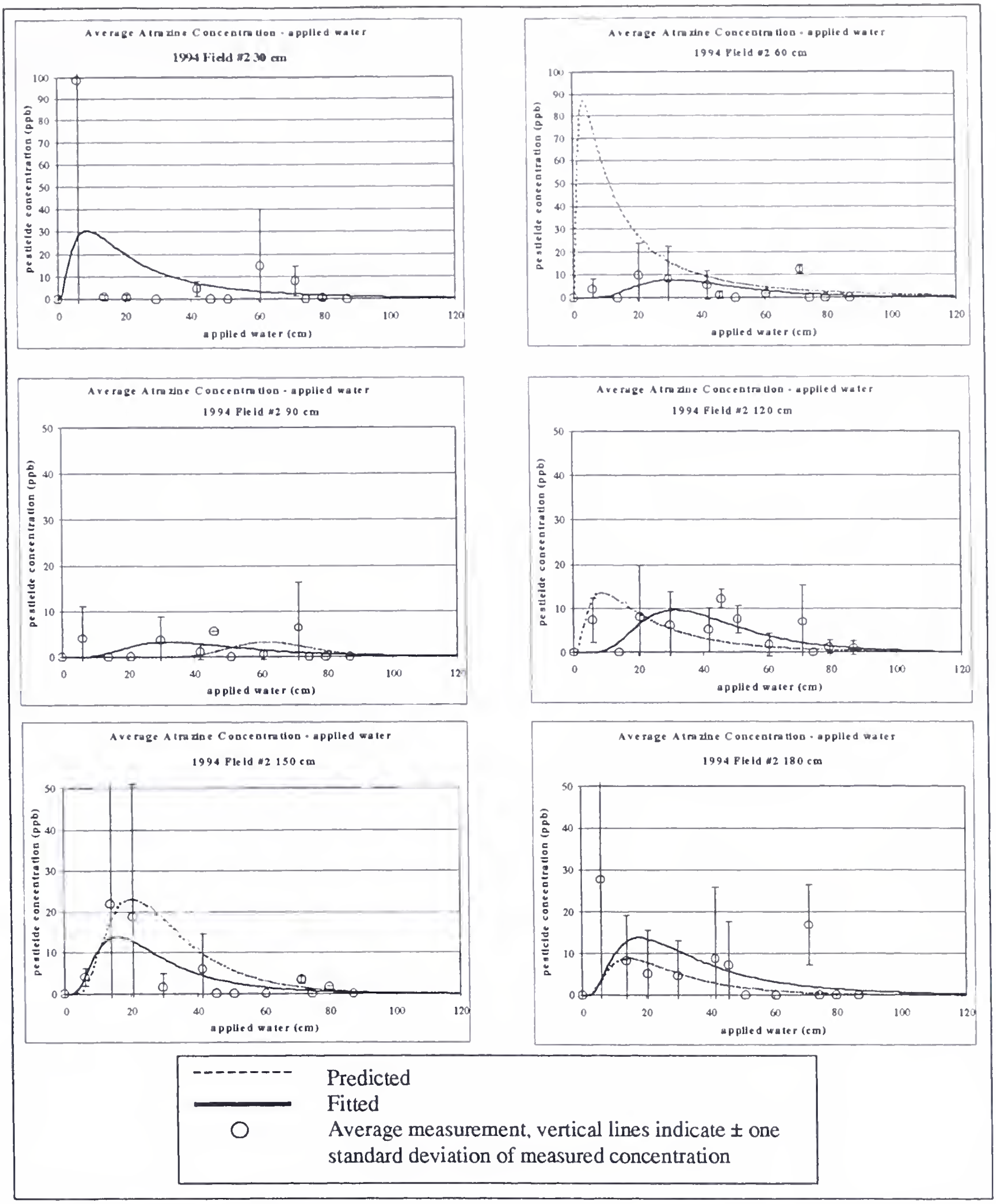

Figure 1. Predictions for Field #2

the standard deviation bars, is also considered. Eventually, a t-test will be used to quantitatively compare the measured and predicted pdf's.

Discussion and Conclusions

One major difference between the 1994 growing season and the two previous growing seasons is the amount of precipitation received. In 1994, a lesser amount of precipitation was received (87.2 cm), compared to the average

amount in 1992 and 1993 (110.5 cm). This represents approximately 22 percent less precipitation and is only partially accounted for by using the 1994 measured concentrations at the 30 cm depth in the calibration. Higher infiltration amounts may, indeed, affect the relationship between pesticide dynamics at the 30 cm depth and lower depths.

Generally, in Field #2, the predicted pdf's could be related to the pdf's fit to the measured data. Some differences may be accounted for by the difference in precipitation and by the fact that samplers were not 100% efficient at capturing the solute.

The graph for the 30 cm depth is shown for reference. Since the 30 cm measured data was used as calibration data to make predictions of solute concentration at lower depths, there is no predicted pdf on this graph.

The predicted pdf for the 60 cm depth has a much higher peak and earlier arrival time (less amount of water) than the pdf fitted to the measured data at the 60 cm depth. The tail of the predicted pdf coincides with the measured data for larger amounts of applied water. Additionally, the mass of pesticide (area under the curve) is much higher for the predicted pdf than for the measured pdf.

For the 90 cm depth, the masses of predicted and measured pesticide concentrations appear to be equal However, the predicted pdf has a later arrival time for peak concentration. The predicted pdf does not fit the measured data well.

At the 120 cm depth, the masses of predicted and measured pesticide concentrations appear to be equal. At this depth, however, the predicted pdf has an earlier arrival time. The predicted pdf passes through, or comes close to, the error bars of the measured data, with 3 exceptions. The data that is missed appears to correspond to a second concentration peak.

The arrival time for the predicted pdf for 150 cm almost coincides with the pdf fitted to the measured data. By visual inspection, it appears that the predicted mass is about 50 percent more than the measured mass of pesticide. The predicted pdf fits the measured data well.

The graph for the 180 cm depth also indicates a close arrival time for the predicted and measured pdf's. However, the predicted mass is about 40 percent less than the measured mass, by visual inspection.

For Field #2, it seems the method used to predict concentrations at lower depths did not perform well at the shallow 60 cm depth. For the intermediate depths of 90 cm and 120 cm, it performed well when predicting pesticide mass and did not do well predicting pesticide peak arrival time. For the lower depths of 150 cm and 180 cm, the method performed well when predicting pesticide peak arrival time. The performance of the method for predicting the pesticide mass at the lower depths was reasonable considering field heterogeneity.

One may note that the stochastic method developed in this study is providing reasonable predictions of pesticide transport to lower depths. Therefore, we may use deterministic models, such as GLEAMS, for shallow depths and a stochastic approach for the lower depths in order to predict pesticide transport for the entire soil profile. We could also speculate that the stochastic approach used in this study was not considering macropore flow, thus it was more susceptible to failure at the 60 cm depth where macropore flow is more likely to occur.

In summary, we concluded that there is great promise for accuracy and feasibility in using such a stochastic approach in predicting pesticide transport through the vadose zone.

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Predicting Atrazine Leaching under Field Conditions Using MACRO Model[1]

Abstract

Field studies indicate that the preferential water movement can allow the surface-applied agricultural chemicals to transport rapidly through the root zone and into shallow groundwater. This paper presents the application of a physically based two-domain model of water flow and solute transport (MACRO) to field condition with a sandy loam soil. One important feature of the MACRO model is that it can be run in one or two flow domains, macropores and micropores. This allows a quantitative evaluation of the impact of macropore flow on solute transport processes. The objective of this study was to quantify the significance of macropore flow in pesticide transport in the Coastal Plain Soils by simulating the MACRO model in both one domain (i.e. without macropore flow) and in two domains (i.e. with macropore flow) and compare the model simulations with field observations. The experimental site is located in the Coastal Plain physiographic region and the dominant soil is Monmouth fine sandy loam with 2-5 % slope. There are four fields. Fields 1 and 2 have been in no-till and fields 3 and 4 have been in conventional-till since 1989. Each year, atrazine was applied at the rate of 1.7 kg/ha to all four fields. Suction lysimeters have also been installed at six different locations in each field, at 30, 60, 90, 120, 150 and 180 cm below the soil surface.

First, a thorough sensitivity analysis was performed in order to identify the most sensitive input parameters. Second, the model was calibrated with 1992 field 2 data. Finally, the model simulations of atrazine concentrations in 1993, 1994 and 1995 were compared to the 95% confidence interval of the lysimeter concentration data. The results indicated that the MACRO model was able to reproduce the observed pattern of the atrazine leaching found in a sandy loam at the shallow depth in 1993 and 1995, providing by-pass flow in soil macropores was considered. However, the model with one domain failed to reflect this pattern. The model also can predict the atrazine concentration at deep depth (depths greater than 90 cm) successfully and there were no significant differences between the model predictions in one domain and two domains due to the lack of macropores at deep depth. About 80% (average) of the model predictions of three years fell within the 95% confidence intervals of the measured data. This study concluded that macropore flow plays an important role in the water movement and solute transport in the Coastal Plain Soils, thus the models with macropore capability such as the MACRO model should be used for pesticide screening instead of Darcian-based models.

Keywords: modeling, MACRO, preferential flow

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Introduction

Field studies indicate that the preferential flow can allow the surface-applied agricultural chemicals to transport rapidly through the root zone (Jabro et al., 1994). The MACRO model used in this study is a comprehensive physically based model describing the water balance, solute transport, and solute transformation processes in the soil/crop system (Jarvis, 1994). This model can be run in one or two flow domains. This capability allows a quantitative evaluation of the impact of macropore flow on solute transport. The objective of this study was to quantify the significance of macropore flow in pesticide transport in the Coastal Plain Soils by simulating the MACRO model both in one domain and in two domains and compare the model simulations with field observations.

Materials and Methods

Field Study and Measurements

The experimental site consisting of four fields each with an area of about 0.2 ha is located in the Coastal Plain physiographic region. The dominant soil is Monmouth fine sandy loam with 2-5 % slope. The average annual rainfall is about 1000 mm. Fields 1 and 2 have been in no-till and fields 3 and 4 have been in conventional-till since 1989. The atrazine was applied at the rate of 1.7 kg/ha. Suction lysimeters were installed at six different locations in each field, at 30, 60, 90, 120, 150 and 180 cm below the soil surface (Vyravipillai et al., 1994). The means of atrazine concentrations at each depth for each sampling date were compared with the model's predictions.

Modeling: Sensitivity Analysis, Calibration, and Validation

The typical input parameter values used in the model sensitivity analysis were selected based on the MACRO model's technical description and sample simulation (Jarvis, 1994). The input parameter values were then increased or decreased by 10%, 25%, 50%, 75%, and 100% of the typical values. After sensitivity analysis, the model was calibrated by changing the values of the most sensitive input parameters until the good fit was obtained between the model's predictions and the measured data. The field 2 data of 1992 was used in the model calibration. Validation of the MACRO model was performed using the field 2 data from 1993 to 1995. When the model runs in two domains, the weighted atrazine concentration in the soil water was calculated based on the concentrations and volumetric water content in micropore and macropore domains, respectively. The model predictions are considered accurate and acceptable if the predicted data fall within the 95% confidence intervals (two standard errors of the mean) of the observed data (Jabro et al., 1994) using statistical and graphical methods.

Results and Discussion

Table 1 shows the most sensitive parameters and their values. Figs 1 to 3 show the model validations from 1993, to 1995. In 1993, the model underestimated the atrazine concentrations at 30 cm depth before the middle of July (Fig. 1). Then the model with two domains overestimated the atrazine concentrations until the end of the year. The model predictions in one domain and two domains were very close before the atrazine application date (5/26/93). This may be due to low predicted atrazine concentrations in the macropores before the application date. The model with two domains was able to mimic the peak time, but it underestimated the peak concentration. The model with one domain failed to predict the peak time and concentration. At 60 cm depth, the model underestimated the atrazine concentrations on most days. The model performed reasonably well at 90, 120, 150, and 180 cm depths for most days. Only small discrepancies existed between the model predications in one domain and two domains at these depths because of few macropores available at deep depth. The model predictions fell within the 95% confidence intervals 7 out of 9 (78%) and 8 out of 9 (89%) in two domains and one domain, respectively, at 30 cm depth. At 60, 90, 120, 150, and 180 cm depths, the model predictions fell within the 95% confidence intervals 5 out of 7 (71%), 5 out of 7 (71%), 7 out of 7 (100%), 8 out of 9 (89%), and 10 out of 12 (83%), respectively.

In 1994, the model failed to predict the peak time and concentration at 30 cm depth (Fig. 2). However, there was a doubt about the measured peak concentration. At 60 cm depth, the model underestimated the atrazine concentrations. The model predictions were close to the measured data at 90, 120, 150, and 180 cm depths, respectively, for most days. There were no significant differences between the model predictions in one domain and two domains at these depths due to the lack of macropores at deep depth. The model predictions fell within the 95% confidence intervals 3 out of 4 (75%) and 1 out of 4 (25%) in one domain and two domains, respectively, at 30 cm depth. But, there was too little data to validate the model performance at this depth using the statistical method. At 60 90, 150 and 180 cm depths, the model predictions fell within the 95% confidence intervals 8 out of 8 (100%), 5 out

Table 1. Six most sensitive input parameters and their values

Parameter	Value
λ: pore size distribution index in the micropore	1.17(1-2)[a], 0.91(3-4), 0.78(5-6), 0.6(7-9), 0.5(10-15)
n*: pore size distribution index in the macropore	5(1-3), 4(4-9), 2(10-15)
f*: impedance factor	1(1-15)
μ (1/d): degradation rate coefficient	0.03(1-3), 0.02(4-9), 0.01(10-15)
C_f: correction factor for the wet canopy evaporation	1(1-15)
K_d (cm^3/g): sorption distribution coefficient	0.87(1-3), 0.75(4-6), 0.5(7-9), 0.29(10-12), 0.174(13-15)

a: (1-2) shows that the value is for layer 1 to layer 2. The soil profile was divided into 15 layers in this study.

of 5 (100%), 6 out of 6 (100%) and 7 out of 8 (88%), respectively. The model simulations fell within the 95% confidence intervals 5 out of 7 (71%) and 4 out of 7 (57%) in one domain and two domains at 120 cm depth.

In 1995, the model responded to the peak time very well but failed to predict the peak concentration at 30 cm depth (Fig. 3). The model overestimated the atrazine concentration greatly at this depth during the growing season. At 60 cm depth, the model with two domains captured the peak time but overestimated the peak concentration. The model with one domain was not able to predict the peak time and value. At 90 and 180 cm depths, there were no enough data to validate the model by using the statistical method. The model underestimated the atrazine concentrations but still performed well at 120 and 150 cm depths. The model predictions fell within the 95% confidence intervals 2 out of 8 (25%), 9 out of 9 (100%), and 5 out of 6 (83%) at 30, 120, and 150 cm depths, respectively. At 60 cm depth, the model predictions fell within the 95% confidence intervals 4 out of 5 (80%) and 1 out of 5 (20%) in one domain and two domains, respectively.

One possible explanation of the model underestimating the atrazine concentration at deep depth is that the MACRO model assumes an instantaneous equilibrium between pesticide in sorbed and solution phases. Kinetic or non-equilibrium sorption would lead to more rapid pesticide transport to the deep depth in the soil profile (van Genuchten and Wagenet, 1989). The model overestimated the atrazine concentrations in the soil water at the shallow depth during the growing season. This may be related to the temporal variation in hydraulic properties of the soil. Meesing and Jarvis (1993) reported that near-saturated hydraulic conductivity decreased significantly during the growing season, and Leeds-Harrison et al. (1986) showed a decrease in K_s with time in clay cores. In the MACRO model, however, the hydraulic parameters were kept constant during the whole simulation.

Conclusions

1) The two-domain MACRO model was able to reproduce the observed pattern of the atrazine leaching found in a sandy loam at the shallow depth in the Coastal Plain Soils. 2) The model also can predict the atrazine concentration at deep depth (depths greater than 90 cm) successfully and there were no significant differences between the model predictions in one domain and two domains due to the lack of macropores at deep depth. 3) About 80% (average) of the model predictions of three years fell within the 95% confidence intervals of the measured data. 4) The macropore flow plays an important role in the water movement and solute transport in the Coastal Plain Soils, thus the models with macropore capability such as the MACRO model should be used for pesticide screening instead of Darcian-based models.

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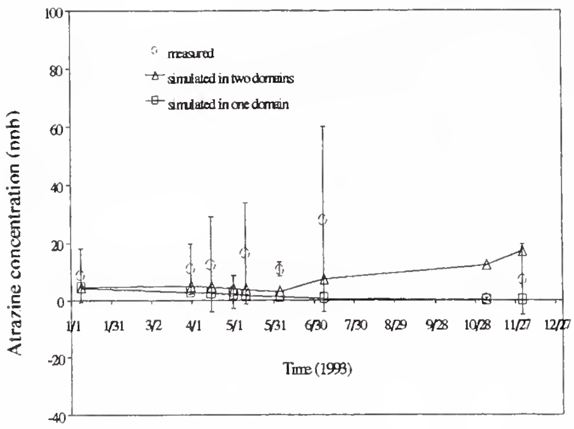
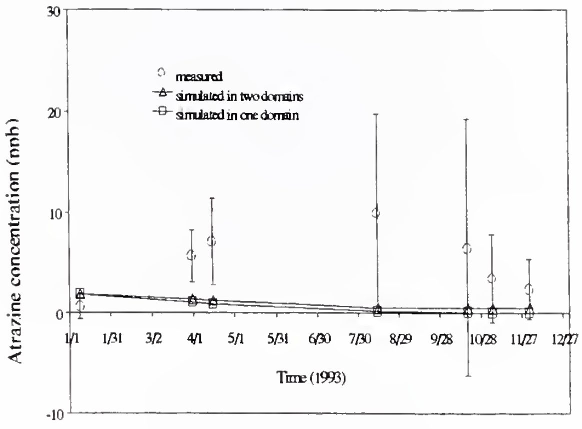


At 30 cm depth



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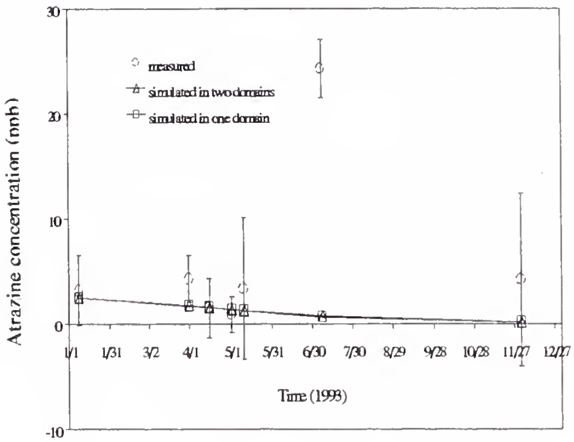

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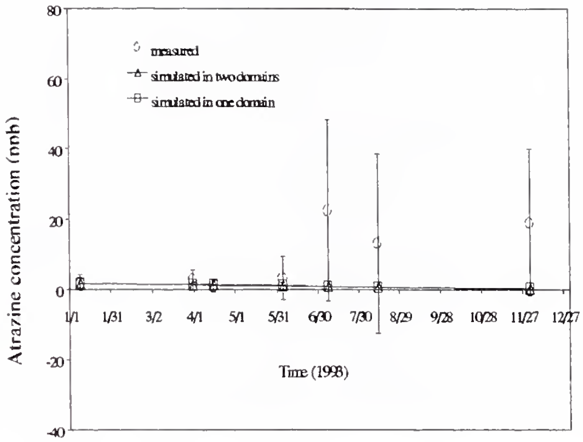

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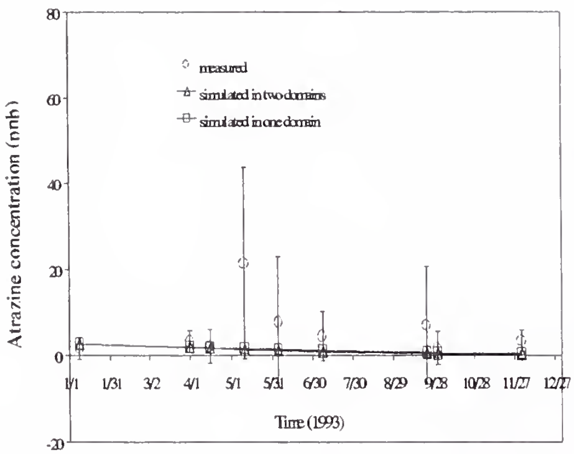

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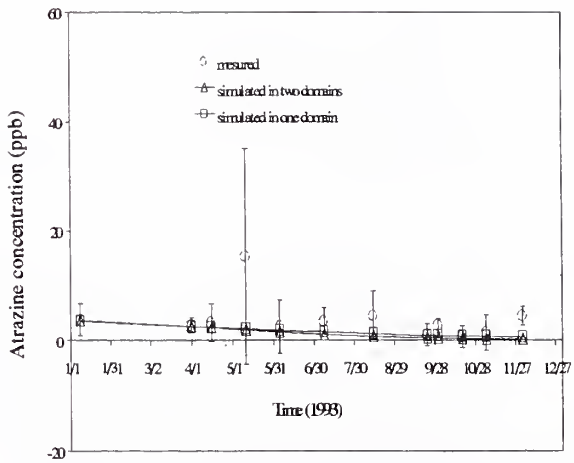


At 180 cm depth

Fig. 1. Comparison of the simulated atrazine concentration in the soil water with the 95% confidence intervals of the field data at different depths in 1993.

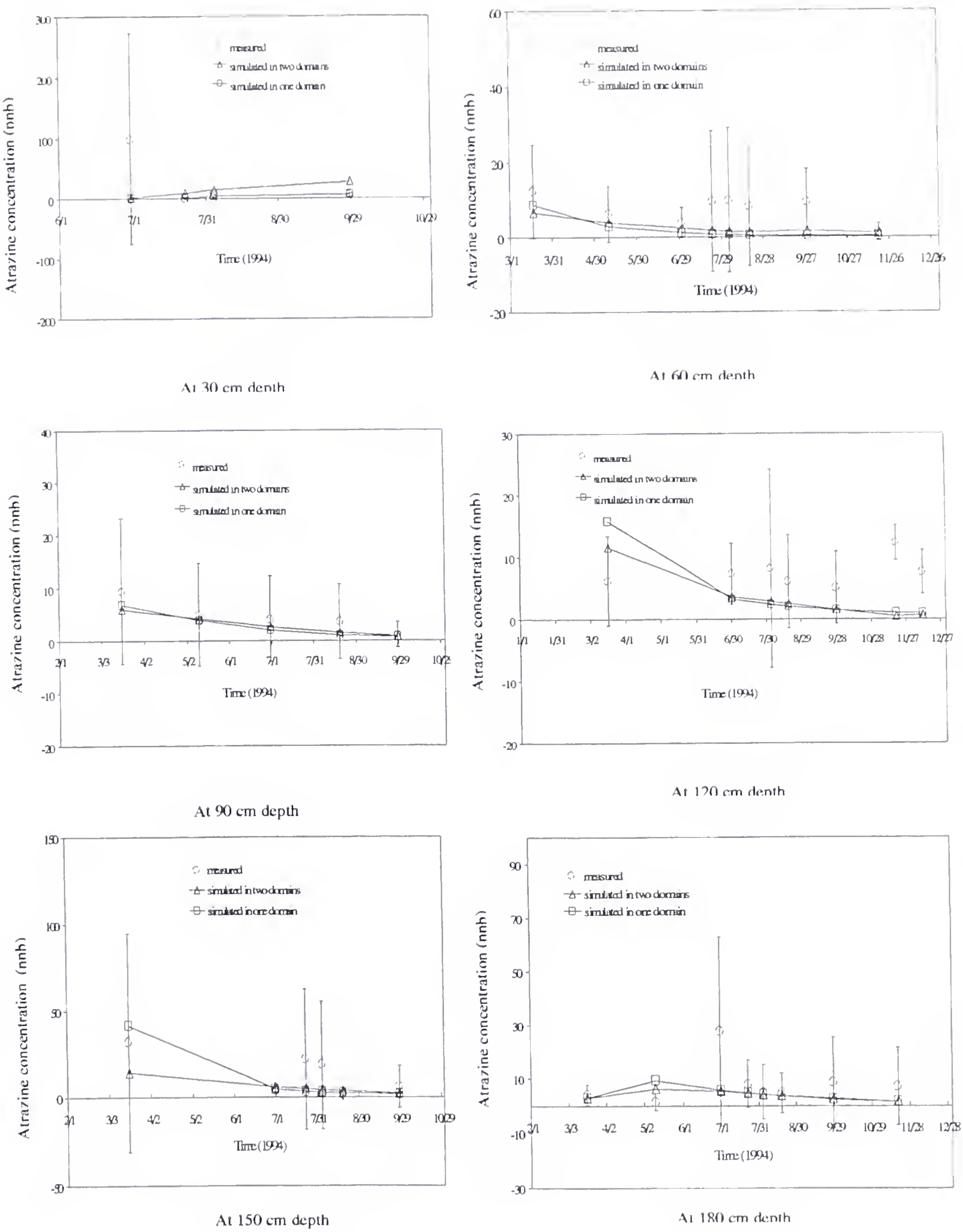
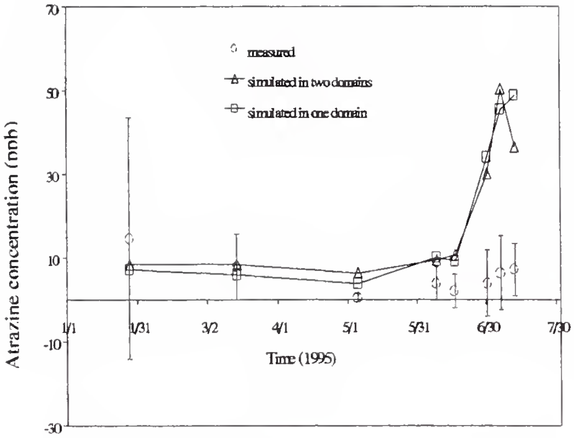
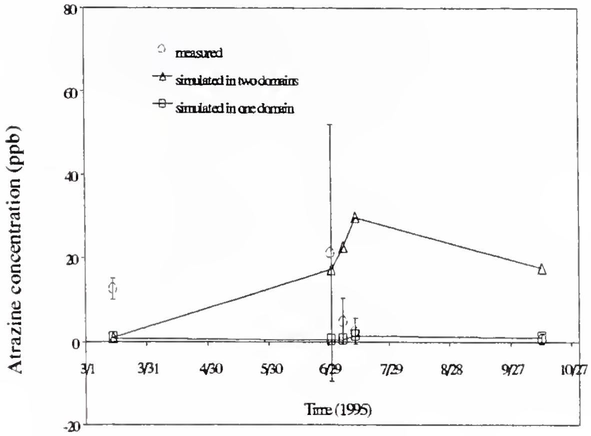


Fig. 2. Comparison of the simulated atrazine concentrations in the soil water with the 95% confidence intervals of the field data at different depths in 1994.



At 30 cm depth



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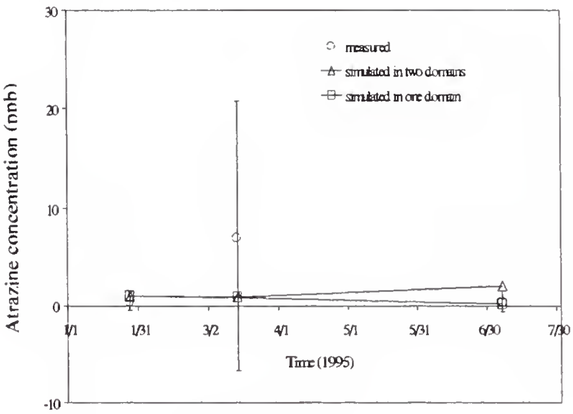

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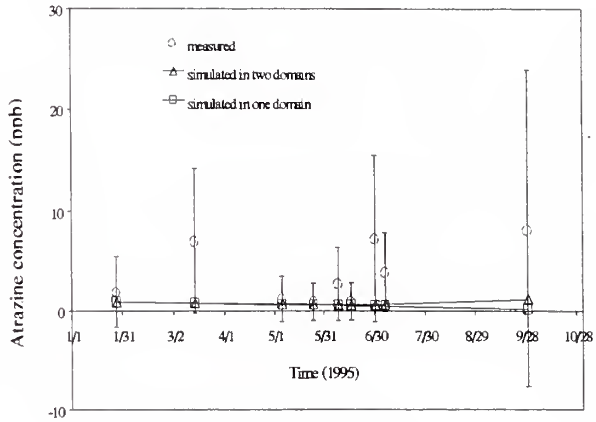
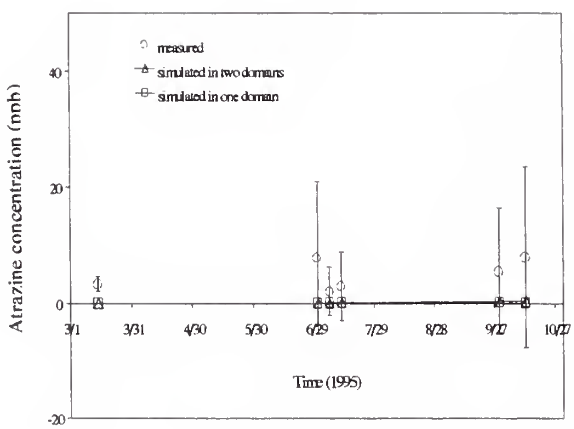
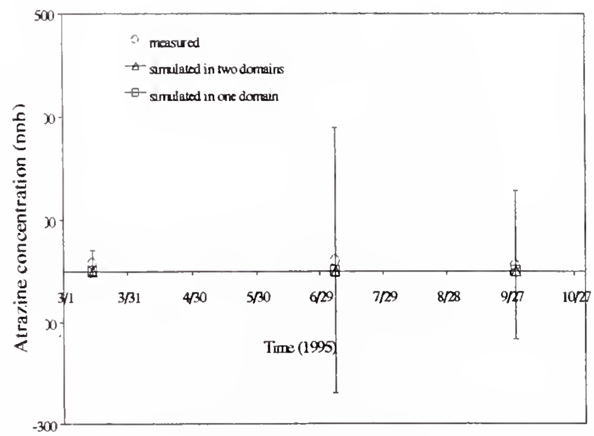


At 120 cm depth



At 150 cm depth



At 180 cm depth

Fig. 3. Comparison of the simulated atrazine concentration in the soil water with the 95% confidence intervals of the field data at different depths in 1995.

Feasibility of EPIC to Assess Long-Term Water Quality Impacts for Fast Growing Herbaceous and Woody Biomass Production [1]

Abstract

The ability of EPIC (Environmental Policy Integrated Climate) to assess the long-term impacts of switchgrass (Panicum virgatum), cottonwood (Poplus spp.), sweetgum (Liquidambar styraciflua) and sycamore (Platanus occidentalis) production systems on runoff quality was investigated. EPIC predicted runoff reasonably well for all tested crops. Twenty-year runoff simulations for sycamore and cottonwood plots showed 31% and 37% less runoff than for no-till corn and cotton plots, respectively. The average magnitudes of predicted and measured TSS discharges from woody plots were small. Twenty-year TSS simulation for woody crops showed no TSS discharges, indicating that TSS discharges from these plots were negligible compared to agricultural production plots. The average magnitudes of predicted and measured NO_3-N and T-P losses from woody plots were small compared with agricultural crops. Twenty-year NO_3-N simulations showed that woody plots had the lowest NO_3-N losses. However, use of EPIC to assess the long-term effect of T-P losses from biomass crops must be exercised carefully.

Keywords : EPIC, water quality, runoff, biomass crop

[1] ASAE Paper 99-2068. J.D. Choi, Assistant Professor, Dep. Of Ag. Eng., Kangwon National Univ., Chunchon Korea 200-701; B.A. Engel, Professor, Dep. Of Ag. and Bio. Eng., Purdue Univ., West Lafayette, IN 47907; V.R. Tolbert, Oak Ridge National Lab., Oak Ridge, TN; B.R. Bock and F.C. Thornton, Tennessee Valley Authority, Muscle Shoals, AL; W. Bandarayanake, D.D. Tyler and A. Houston, West Tennessee Experimental Station, Univ. Of Tennessee, Jackson, TN; D. Pettry and S. Schoenholtz, Mississippi State Univ., Stoneville, MS; T.H. Green, Alabama A&M Univ., Normal, AL. Contact author (jdchoi@cc.kangwon.ac.kr).

Introduction

Biomass resources have been historically important for energy supplies and considered as an alternative to fossil fuels. Biomass power generation in the U.S. has grown from 200 MW in 1979 to more than 6,000 MW in 1990. The U.S. DOE (Department of Energy) forecasts up to 20 GW of electricity generation by the year 2010. TVA (Tennessee Valley Authority) and the U.S. Department of Energy's Bioenergy Feedstock Development Program (BFDP) are investigating options to incorporate biomass fuel into TVA's near- and long-term power generation operations which could supply up to 2,000 MW of biomass base load capacity by 2010. To achieve this level of biomass fueled power generation, it is anticipated that approximately 405,000 ha of woody biomass production would be required (Downing and Graham, 1993). Wright and Hughes (1993) reported that the northeast, south central and southeastern states of the U.S. are the most suitable for the production of fast growing energy crops.

Fast growing herbaceous and woody energy crop production systems have been investigated under BFDP in three physiographic regions of the southeastern U.S. The effects of the conversion of land from conventional food and fiber crop cultures to fast growing biomass energy crops on water quality have been experimentally investigated (Tolbert et al., 1997; Houston, 1996; Thornton et al., 1997; Green et al., 1996; Pettry, 1997). Ranney and Mann (1994), and OTA (1993) have also described the effects of biomass production on environmental quality. However, both quantitative field and modeling studies on the long-term effects of biomass production systems on water quality at the edge of field using monitored data are limited. Field monitored data sets of runoff, total suspended solids, nitrate-nitrogen and total phosphorus on the fast growing herbaceous and woody biomass production systems collected by BFDP are needed to model to predict the long-term water quality impacts of the system. And, the use of models to examine the potential water quality benefits of biomass production is of interest.

EPIC (Environmental Policy Integrated Climate formerly Erosion-Production Impact Calculator) has been applied to numerous projects related to agricultural production, soil erosion and environmental quality, and publications and references about EPIC were well documented (Mitchell, 1998). The objective of this study was to investigate EPIC's ability to assess the long-term impacts of fast growing herbaceous and woody crop production systems on water quality at the edge of fields by comparing monitored and predicted data. The analysis of EPIC's ability to estimate these parameters will be an indication of its ability to predict the long-term environmental impacts of the fast growing bioenergy crops on water quality. The results may be used to help develop environmentally sound biomass production systems and predict the long-term water quality impacts of the systems.

Methods

EPIC was developed to assess the effect of soil erosion on productivity and predict the effects of management decisions on soil, water, nutrient and pesticide movement and their combined impact on soil loss, water quality and crop yields for areas with homogeneous soils and management. Many model parameters are either readily available from data bases or chosen from options within the model interface. Among six daily weather inputs, daily rainfall was collected at the experimental sites and other weather inputs were either measured or simulated depending on the site records. For runoff and soil erosion, the SCS CN method and the small watershed version of MUSLE (Mitchell et al., 1998) options were chosen, respectively. Soil and other site specific and management data were provided based on the field data from the individual study sites.

The three experimental sites and runoff plot configuration and treatment are shown in Table 1. An earth berm about 50 cm in height was constructed around each plot to hydrologically isolate the plots from surrounding fields, and H-flume systems were set up to measure rainfall event based runoff volume, total suspended sediment (TSS), nitrate-nitrogen (NO_3-N), and total phosphorus (T-P) leaving each plot. Management practices and fertilizer applications for no-till corn and cotton crops followed typical customary practices around the respective experimental site.

Switchgrass was planted in May, 1995 and no further planting was made throughout the study. Sweetgum trees were planted on 1.5 m within row and 3 m between row spacings. On the sweetgum plots with fescue cover, a 2 m wide fescue crop was established down the center between rows. Sycamore trees were planted at 1.2 m within row

Table 1. Runoff plot configuration and treatments of the three sites.

Site	Plot No.	Size (ha)	Slope (%)	Treatment	Soil
AL A&M University, Hazel Green, AL	1	0.43	3.1	No-till corn (Zea mays L.)	Decatur Silty Clayey Loam
	2	0.43	3.1	Switchgrass (Panicum virgatum)	
	3	0.43	2.9	Sweetgum (Liquidambar styraciflua L.) With fescue (Festuca elatior L.) cover	
	4	0.43	3.1	Sweetgum (Liquidambar styraciflua L.)	
	5	0.18	4.5	Switchgrass (Panicum virgatum)	
	6	0.18	6.0	Sweetgum (Liquidambar styraciflua L.)	
	7	0.18	5.5	No-till corn (Zea mays L.)	
	8	0.18	5.0	Sweetgum (Liquidambar styraciflua L.) With fescue (Festuca elatior L.) cover	
Ames Plantation, Grand Junction, TN	1	0.49	2~3	Sycamore (Platanus occidentalis L.)	Loring-Memphis Silty Loam
	2	0.49	2~3	Sycamore (Platanus occidentalis L.)	
	3	0.49	2~3	Sycamore (Platanus occidentalis L.)	
	4	0.49	2~3	No-till corn (Zea mays L.)	
	5	0.49	2~3	No-till corn (Zea mays L.)	
	6	0.49	2~3	No-till corn (Zea mays L.)	
Delta Res. Center, Stoneville, MS	1	0.36	0.13	Cotton (Gissypium hirsutum L.)	Bosket Silty Loam
	2	0.36	0.22	Cottonwood (Populus deltoides Bartr)	
	3	0.36	0.10	Cottonwood (Populus deltoides Bartr)	
	4	0.36	0.12	Cotton (Gissypium hirsutum L.)	
	5	0.36	0.13	Cottonwood (Populus deltoides Bartr)	
	6	0.36	0.19	Cotton (Gissypium hirsutum L.)	

by 2.4 m between row spacing. Cottonwood cuttings were established at a spacing of 1.2 m within each row by 3.6 m between each row. All tree crops were planted in February and March, 1995. Detailed experimental procedures and some of the monitoring results were reported by Tolbert et al. (1997).

The evaluation of EPIC's ability to predict the measured variables was made by performing graphical comparisons between measured and predicted variables and analyzing the coefficient of determination and the Nash-Sutcliffe Efficiency (Nash and Sutcliffe, 1970). Twenty-year long-term simulation for each crop also was performed to examine the trends of measured variable changes over time with respect to the respective control crop which was no-till corn or cotton.

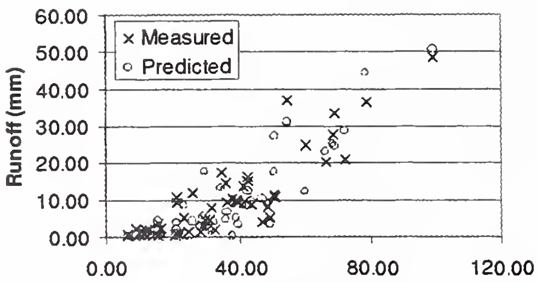
Results and Discussion

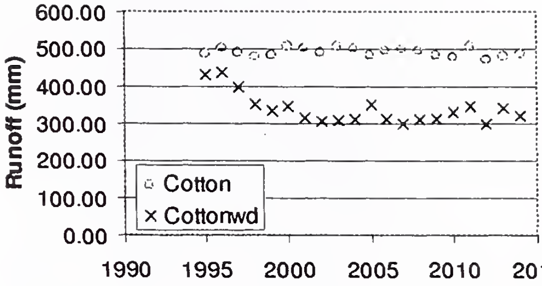
Runoff

Measured runoff from no-till corn and sycamore plots at the TN site and from cotton and cottonwood plots at the MS site were not significantly different ($\alpha = 0.05$) during the 3-year monitoring period. Measured runoff at the AL site showed significant difference ($\alpha = 0.05$) between 0.49 ha and 0.18 ha plots. Small (0.18 ha) plots produced more runoff than large (0.49) ha plots. Measured runoff from four treatments within the same size plots at the AL site did not show significant differences ($\alpha = 0.05$) during the 2-year monitoring period. However, measured runoff at the AL site showed very high variation.

Predicted runoff by EPIC at the TN and MS sites fit reasonably well with measured runoff (Figure 1). The coefficients of determination and the Nash-Sutcliffe Efficiencies between measured and predicted runoff showed relatively high association (Table 2). Predicted runoff by EPIC at the AL site was not as good as the other two sites. A relatively short monitoring period (2 years), no replication, and large variations in measured runoff might cause the low coefficients for the AL site.

Twenty-year long-term runoff simulations with a fixed annual rainfall for the TN and MS sites showed significantly reduced runoff from sycamore and cottonwood plots compared to no-till corn and cotton plots.





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sycamore plots at the Ames Plantation site, Tennessee.

1 a

fixed rainfall of 1,606 mm for cotton and cottonwood plots at the Delta Research Center site, Mississippi.

Table 2. Coefficient of determination (R^2) and Nash-Sutcliffe Efficiency (N-S) between measured and predicted runoff.

Site	AL A&M University Hazel Green, AL (0.49 ha plot)				Ames Plantation, Grand Junction, TN		Delta Research Center Stoneville, MS	
Crop	Corn	Switch-grass	Sweetgum with cover	Sweet-gum	Corn	Syca-more	Cotto n	Cotton-wood
R^2	0.75	0.67	0.28	0.37	0.74	0.75	0.61	0.71
N-S	0.04	0.59	-0.23	-0.18	0.67	0.72	0.59	0.71

respectively (Figure 2). The fixed annual rainfalls for the two sites were 1,300 mm (TN site) and 1,606 mm (MS site), based on the measured rainfall in 1995. Runoff simulated from no-till corn and cotton plots remained the same over time; while runoff from sycamore and cottonwood plots gradually decreased with time until stabilization was reached. Annual runoff from sycamore and cottonwood plots after runoff stabilization were about 31% and 37% less than that from no-till corn and cotton plots, respectively. The AL site also showed significantly reduced annual runoff for the sweetgum with fescue cover plots compared to no-till corn plots. However, annual runoff from no-till corn, switchgrass and sweetgum plots did not show significant differences. The conversion from both no-till corn and cotton cultures to fast growing woody crop cultures may significantly reduce runoff and runoff related non-point pollutant discharges, and may contribute to the improvement of local water quality.

Total Suspended Solids (TSS)

Measured TSS discharges from sycamore and cottonwood plots at the TN and MS sites ranged from 0 to 26.0 kg/ha, and 0 to 297.0 kg/ha, respectively, which were relatively small compared to no-till corn and cotton plots where the TSS discharges were 0 to 30.0 kg/ha, and 0 to 2,810.0 kg/ha, respectively. Predicted TSS discharges from sycamore and cottonwood plots at the TN and MS sites did not fit well with measured TSS discharges at the beginning of the study, but both measured and predicted TSS discharges decreased with time (Figures 3). The magnitudes of predicted and measured TSS discharges from both plots for the third year were very small and the differences between them also were very small. Measured TSS discharges from no-till corn and sweetgum plots at the AL site varied widely during the two years of study included in this analysis. Sweetgum plots did not reach stabilization in two years; and no replication of each treatment probably account for the wide variation. The coefficient of determination and the Nash-Sutcliffe Efficiency of the 3 sites were low because the differences between measured and predicted TSS discharges were large at the beginning of the study.

Twenty-year long-term TSS simulation at three sites for sweetgum, sweetgum with fescue cover, sycamore, and cottonwood showed zero TSS discharges after these trees reached canopy closure and site stabilization. Zero TSS discharges may somewhat under-estimate the real TSS discharge from these woody plots but indicate that the magnitudes of TSS discharges from these plots are negligible compared to conventional food and fiber production

plots, indicating that the woody biomass production systems have very little water quality impact compared to traditional crop production systems.

Nitrate Nitrogen (NO_3-N)

Measured NO_3-N in runoff from no-till corn and sycamore plots at the TN site ranged from 0 kg/ha to 2.205 kg/ha and 0 kg/ha to 0.544 kg/ha, respectively. For the MS site, measured NO_3-N losses from both cotton and cottonwood were not significantly different, although cotton plots lost slightly more NO_3-N than cottonwood plots. Nitrogen fertilizer applied to both cotton and cottonwood plots were not much different during the 3-year study and thus, NO_3-N losses from both plots would be expected to be similar. Measured NO_3-N losses generally ranged from 0 to 0.337 kg/ha for cotton plots and 0 to 0.241 kg/ha for cottonwood plots. For the AL site, measured NO_3-N losses from no-till corn and switchgrass plots were greater than those from sweetgum and sweetgum with fescue cover plots. Measured NO_3-N losses were generally less than 0.2 kg/ha from 0.43 ha switchgrass plots and 0.1 kg/ha from 0.43 ha sweetgum and sweetgum with fescue cover plots. However, much higher NO_3-N losses were observed when fertilizer application and large rainfall events nearly coincided. NO_3-N losses from 0.18 ha plots at the AL site were generally greater than those from 0.43 ha plots.

Predicted and measured NO_3-N from sycamore plots at the TN site did not fit well. For the MS site, predicted NO_3-N losses from both cotton and cottonwood plots were generally greater than measured NO_3-N losses. Predicted NO_3-N losses at the AL site also did not fit well with measured NO_3-N losses. Large differences between measured and predicted NO_3-N losses were observed when nitrogen fertilizer application and rainfall events nearly coincided, with EPIC tending to over-predict NO_3-N loss. The differences between predicted and measured NO_3-N losses were very large particularly during the initial years of establishment. The coefficients of determination and the Nash-Sutcliffe Efficiencies were very low because of the large differences.

Twenty-year NO_3-N simulation with a fixed annual rainfall showed that biomass crops released less NO_3-N than no-till corn at the AL and TN sites (Figure 4). Cottonwood plots at the MS site showed the same trend. Predicted NO_3-N losses from sweetgum and sycamore plots showed a stable pattern while those from the switchgrass plot oscillated because of a periodic fertilizer application. Annual variation of predicted NO_3-N losses from the agricultural crop (no-till corn and cotton) plots was greater than that from switchgrass plots.

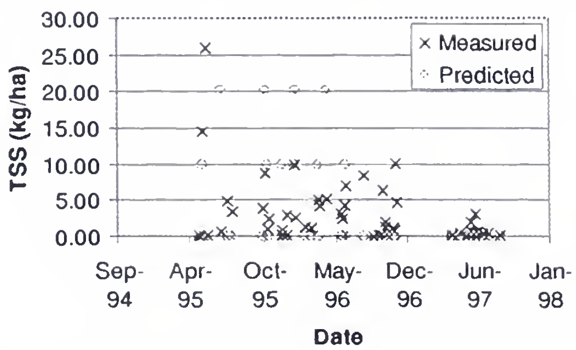

Figure 3. Measured and predicted TSS discharges from sycamore plots at the Ames Plantation site, Tennessee.

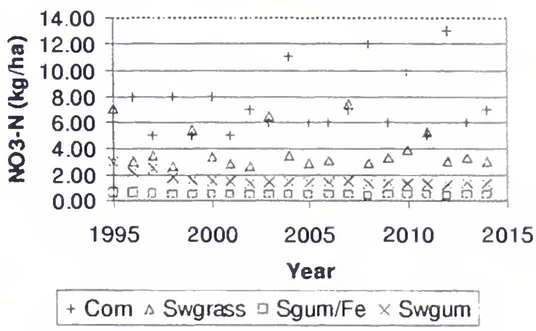

Figure 4. Results of 20-year nitrate-nitorgen simulation with a fixed rainfall of 1,597 mm for no-till corn, switchgrass, sweetgum with fescue, and sweetgum plots at the AL A&M site, AL.

Total Phosphorus (T-P)

Measured T-P losses from sycamore plots at the TN site were affected largely by rainfall, and high T-P losses were measured with large rainfall events that caused relatively large TSS discharges. The same was observed at the other two sites. Predicted and measured T-P losses from biomass crop plots at the three sites did not fit well with each other. Measured T-P losses were usually greater than predicted ones. The coefficient of determination and the Nash-Sutcliffe Efficiency were very low.

Predicted T-P losses from biomass crops were much less than measured T-P losses, indicating that EPIC under-predicted T-P losses. However, the magnitudes of T-P losses from biomass crop plots were small compared to no-till corn and conventional cotton plots. Because of the large discrepancies between measured and predicted T-P losses, long-term simulations to estimate T-P releases from biomass production systems were not performed.

Conclusions

The ability of EPIC to assess the long-term impacts of fast growing herbaceous and woody biomass production systems on water quality at the edge of fields was investigated. Two food and fiber crops and four biomass crops were randomly assigned to field-scale runoff plots at three physiographic regions of south central states of the USA. The runoff plots were monitored from May, 1995 for two or three years. The monitored data sets were modeled with EPIC and long-term simulations were made to evaluate EPIC's ability to assess the long-term impacts of fast growing biomass production systems on water quality with respect to runoff, total suspended solids (TSS), nitrate-nitrogen (NO_3-N) and total phosphorus (T-P).

EPIC predicted runoff reasonably well for all tested crops. The coefficients of determination and Nash-Sutcliffe Efficiencies at the TN and MS sites showed relatively high association between measured and predicted runoff. These coefficients were low at the AL site because of large variations in measured runoff due to a short monitoring period and no replication of the treatment. Twenty-year runoff simulations with fixed annual rainfall for TN and MS sites showed that annual runoff from sycamore and cottonwood plots were 31% and 37% less than that from no-till corn and cotton plots, respectively, after runoff stabilization was reached. Twenty-year runoff simulation for the AL site showed that annual runoff from switchgrass and sweetgum plots was not much different from that from no-till corn plots, but annual runoff from sweetgum with fescue cover plots was significantly less than that from no-till corn plots.

Predicted TSS discharges by EPIC did not fit well with measured TSS discharges for most simulations. However, the magnitudes of predicted and measured TSS discharges from woody plots were not much different and were small compared to the discharges from no-till and cotton and generally decreased with time. Twenty-year TSS simulation for all woody crop plots showed zero TSS discharges after stabilization. This may somewhat under-estimate the real TSS discharge from these woody plots, but the simulations did show that the magnitudes of TSS discharges from these plots are negligible compared to those of conventional food and fiber production plots.

Predicted NO_3-N losses by EPIC did not fit well with measured NO_3-N losses. However, the magnitudes of predicted and measured NO_3-N losses from woody plots were small compared to the discharges from no-till corn and cotton plots. Twenty-year NO_3-N simulations for biomass crop plots showed that woody plots allowed the least NO_3-N loss, followed by switchgrass plots. Predicted T-P losses by EPIC were smaller than measured T-P losses and did not fit well with measured T-P losses. Because of large differences between measured and predicted T-P losses, use of EPIC to assess long-term effect of T-P losses from biomass crops must be exercised carefully.

EPIC showed a reliable ability to predict runoff from fast growing biomass production systems. The conversion from both no-till corn and cotton cultures to fast growing woody crop cultures may significantly reduce runoff and runoff related non-point pollutant discharges from the fields, and may contribute to the improvement of local water quality. However, EPIC's ability to predict NO_3-N and T-P losses from the biomass production systems may need to be improved for better assessment of the systems on environmental quality.

Acknowledgement

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Using an Integrated Model to Evaluate the Hydrological Responses of a Mixed Landuse Watershed

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This paper was not available for inclusion in the proceedings.

Testing of a Watershed Scale Hydrologic/Water Quality Model for Poorly Drained Soils[1]

Abstract

In recent years physically based distributed watershed scale hydrologic/water quality models have been continuously developed, tested and applied to predict impacts of both point and non-point source pollution on receiving waters. Because many of these models have been developed for upland watersheds they are unable to accurately represent the physical processes governing subsurface flow driven by shallow water tables in relatively flat low lying fields. Similarly, these models do not describe the in-stream hydraulics of backwater conditions that occur in the lower coastal plains where slopes are flat and tidal fluctuations may affect the outlet. Although there are models that can describe the processes that occur in the field and outlet separately, there are only a few models that are distributed and comprehensive in nature and are able to adequately describe both processes at the same time.

The main objective of this study was to test a DRAINMOD based, watershed scale, distributed hydrologic model for its performance in predicting hydrology both at the watershed outlet and in-stream locations in a relatively flat coastal landscape. The model uses flow routing procedures, based on numerical solutions of 1-D St. Venant equations in the canal/stream reaches, to predict depth and flow rate. The procedures take backwater conditions into consideration. Nutrient loading or concentration at the edge of a field can be simulated by using DRAINMOD-N, a nitrogen version of DRAINMOD. In other cases, model predicted drainage outflows can be used with a flow-weighted concentration to obtain field loadings at the individual outlets. Depending upon objectives, the model can also be linked with water quality transport and transformation submodels of varying degrees of complexity (from process based in-stream ADR transport and transformation equations to a lumped equation with a first order kinetics) to provide a means of predicting both hydrology and nutrient loadings from these watersheds. Both categories of a comprehensive model will depend upon the flow rates and velocities predicted by the hydrology component of the model.

This study reports results of analyses to determine the model's capability to simulate the daily outflows at the watershed outlet, field hydrology (water table in the field and flow rates at the field edge) and in-stream hydraulics using limited calibration data. The model was applied on two different drained subwatersheds. One is a managed pine forest (2950 ha) and the other is primarily an agricultural watershed (710 ha). Data available in the literature were used as input to the model. Given the complexity of the model and limited data, predictions of daily and annual outflows at the outlets of both subwatersheds were acceptable. The model performed better for the uniform managed forest than for the agricultural watershed, which was more heterogeneous with respect to soils, crops, and water management practices. Using the same general input data from the literature, the model's predictions of daily hydrology of an individual field within the watershed showed poorer performance compared to the predictions obtained by using measured soils data for that field. The results showed that intensive calibration and validation of the hydrologic model with multi-response data like water table depths are needed to accurately quantify the hydrology, in-stream hydraulics, and nutrient loadings at both the field and watershed outlets on a finer time scale. However, the results from this and another companion study indicated that with only minor calibration, the model, linked with lumped parameter water quality submodels, can be a useful tool for evaluating annual cumulative impacts of management practices on large poorly drained watersheds.

Keywords: Watershed Scale Modeling, DRAINMOD, In-stream hydraulics, Nutrient Loadings, Water Quality.

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Introduction

Due to advent of high speed computers along with the recent advancements in hydrological sciences and in-stream monitoring technologies, comprehensive distributed hydrologic and water quality models are being developed for describing impacts of field and stream processes on downstream receiving waters. However, most of these studies have been conducted in the upland watersheds with different hydrologic and in-stream hydraulic processes compared to the relatively flat, poorly drained soils of the lower coastal plains. Fernandez et al. (1997) linked DRAINMOD-N (Breve et al., 1997) with a Dutch in-stream hydraulic model to describe the hydrology and water quality of large coastal watersheds. Preliminary results of simulations predicting nitrogen (N) loads on monthly and seasonal time scales using a newly developed GIS based lumped parameter water quality model are being presented in the proceedings of this conference (Fernandez et al., 1999). In that study, field loadings were based on measured flow rates and concentrations for some individual fields. Loadings for the other fields were interpolated from these experimental results. Transport of N in the drainage canal system was modeled using travel times obtained in the numerical simulations of the DUFLOW model. The computed annual N loadings at the watershed outlet were in good agreement with measured data for a two-year period. Similarly, a DRAINMOD (Skaggs, 1978) based watershed scale hydrologic model with an in-stream flow routing component has been successfully tested in different coastal watersheds of North Carolina (Konyha and Skaggs, 1992; Amatya et al., 1997; Amatya et al., 1998). Recently, Amatya et al. (1999) presented procedures for simulating annual N loads using export coefficients and delivery ratios. Export coefficients were computed by multiplying measured flow weighted N concentrations from individual fields by annual outflows predicted by a DRAINMOD based watershed scale hydrologic model, DRAINWAT. Delivery ratio, a fraction of the N loading delivered from each field to the watershed outlet, was computed using time of travel simulated by DRAINWAT in the first order decay equation. The main objective of this study is to test the performance of DRAINWAT using limited input data for poorly drained watersheds with diverse land management practices. The second objective is to test the model's ability to reproduce individual field hydrology (outflow rates and water table depths) and in-stream flow rates. This is needed because (a) the initial nutrient loading at the field edge (an input to the in-stream transport submodel) is simulated by DRAINMOD-N based on the water table and flow rates computed by DRAINWAT, and (b) the mechanistic or lumped water quality submodels for nutrient transport processes use the velocities and flow rates also predicted by the in-stream flow routing component of DRAINWAT. In other words, this would allow to check the internal consistency of the distributed model and the accuracy of input data in general, as argued by Ambroise et al. (1995).

Site Description and Methodology

The hydrology component of the model was tested with data from two subwatersheds of our 10,000 ha watershed study site near Plymouth, North Carolina (Fig. 1). The site, located on relatively flat (< 0.1 % slope), poorly drained, high water table soils, is basically comprised of managed forests and agricultural lands. The primary drainage system in both watersheds is a network of field ditches (90 to 100 cm depth with 80 to 100m spacing) and canals, which divide the watershed into a mosaic of regularly shaped fields and blocks of fields. The first subwatershed (S4) is an area of about 2950 ha. It drains mainly managed pine forest stands and some mixed pine and hardwood. The surface vegetation varies from field to field and ranges from unharvested second growth mixed hardwood and pine forest to loblolly pine plantation (*Pinus taeda L.*) of various ages and stages (Weyerhaeuser Company, unpublished data). The second subwatershed (T4) is about 710 ha in area and primarily drains intensively farmed agricultural lands. A 137 ha of natural wetland is located in the upstream part of this subwatershed. Corn-wheat-soybean is a regularly planted primary crop on these lands. Flow monitoring stations are located along the drainage canals and at the outlets of the experimental fields, and at the subwatershed outlets. Continuous flow monitoring has been conducted since early 1996. The outlet of the watershed S4 is a dual span 120° V-notch weir draining to a 2.1 m diameter CMP culvert. The subwatershed T4 drains through an open canal into a dual concrete box culvert. Complete weather data from two automated stations, one within S4 and another within T4 is being used to estimate potential evapotranspiration (ET). There are three rain gauges (R1, R6, and R8) within S4 and two gauges (R2 and R3) within the T4 subwatersheds that can be used to estimate aerially distributed rainfall. Water table depths have been continuously measured in wells midway between parallel ditches in experimental fields in both subwatersheds. Both mineral and organic soils are present in these subwatersheds (SCS, 1981). The detailed description of the site and experimental procedures have been outlined elsewhere (Chescheir et al., 1998; Amatya et al., 1998).

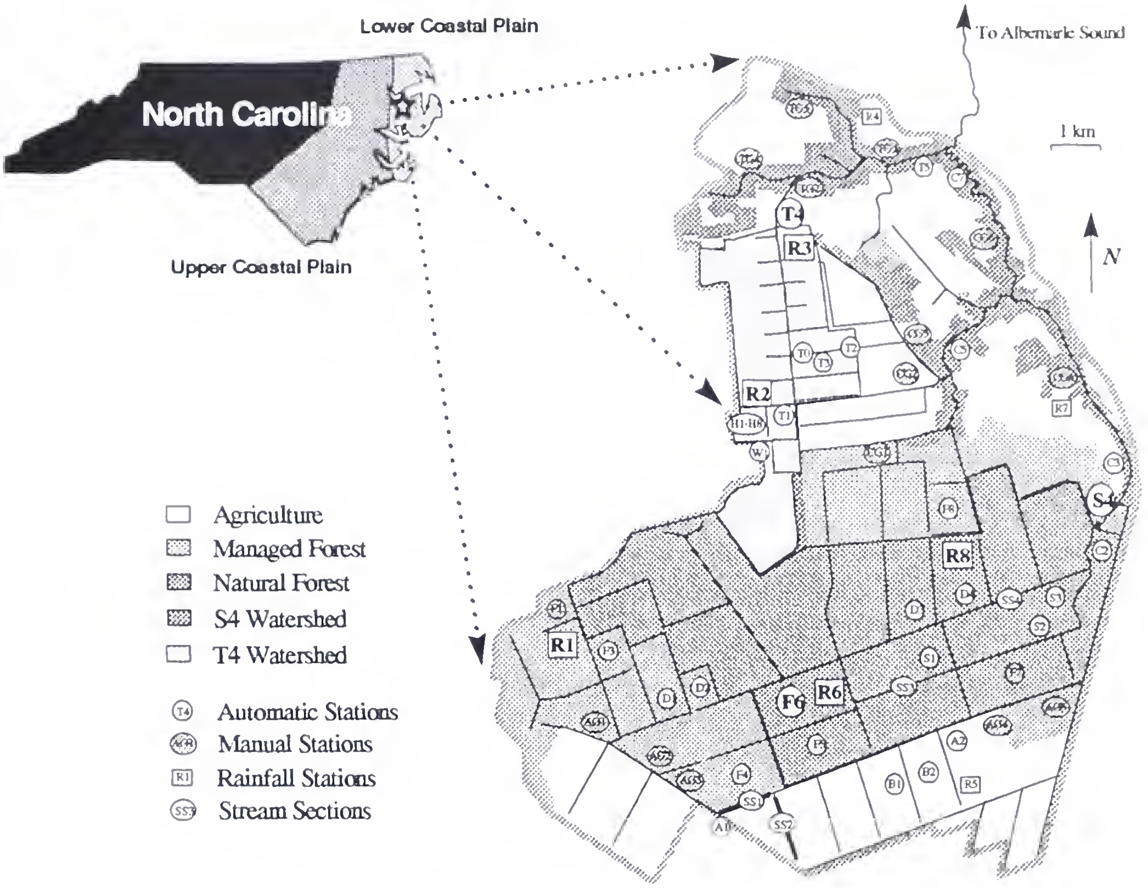


Figure 1. Schematic diagram of 10,000 ha watershed with location of two study sites S4 and T4 (not to the scale).

For modeling purposes the subwatershed S4 was discretized into 27 fields with varying areas (42 ha to 205 ha). Subwatershed T4 was divided into 30 fields with areas varying between 10 ha to 79 ha. Each of the individual fields was discretized based on common drainage, soils and vegetation management practices. The areas of the fields were obtained from geographic information system (GIS) data based on digital orthophotos with ground verification. Measured soil water properties and crop rooting depths for most of the fields were not available and were estimated based on data collected in past studies (Evans, 1991; Chescheir et al., 1995; Breve et al., 1997; Amatya et al., 1998). Constant average effective surface storages of 1cm, 4 cm, and 10 cm were assumed for agricultural fields, wetland, and all forested fields, respectively. Rainfall from the gauges at R6 and R2 were used as input to the model for S4 and T4, respectively. Canopy interception for forests was not modeled. Daily Penman-Monteith REF-ET for the forest and grass references calculated using weather data from the forest and agricultural sites, respectively, were used in the model. Time of concentration was calculated using the procedures described by Amatya et al. (1998). The collector ditch and canal network was divided into stream segments with 50 nodes, 7 branches and 7 weir control structures for routing the flows to the watershed outlet at S4 . Similarly, 69 nodes, 11 branches, and 4 in-stream weirs were simulated in subwatershed T4. A uniform slope of 0.0001 was assumed for these very flat lands. Manning roughnesses of 0.025 and 0.035 were assumed for collector ditches/canals in agricultural lands and forested lands, respectively. Dimensions and elevations of these ditches and weir control structures were also obtained from the GIS and ground level surveys (Unpublished data). Details of the model and the modeling procedures are described elsewhere (Konyha and Skaggs, 1992; Amatya et al., 1997). Using minimum calibration, the model was tested on both S4 and T4 with less than 3 years of measured flow data.

Simulation Results

The daily drainage rates predicted by the model for the forest subwatershed S4 were in good agreement with measured data for all three years (Fig. 2). Most of the model overpredictions occurred during large events on days 262, 283 in 1996 (tropical storms) and day 765 in 1998 (winter event) when the weirs were submerged. Some other discrepancies during the summer and fall were attributed to potential errors in both rainfall and estimates of PET. Note that the predictions were obtained by using rainfall from only one station in the watershed. Relatively large differences in annual rainfall were observed between gauges at R1, R6, and R8 (Amatya et al., 1998); therefore, it seems reasonable that a part of the difference could have been due to the use of the R6 rainfall data for the entire watershed. The average absolute daily deviations between the measured and predicted flows for each of the years 1996, 1997 and 1998 were 0.56, 0.18 and 0.21 mm/day, respectively. Similarly, the correlation coefficients (r) between the measured and predicted daily flows for these three years were 0.89, 0.90 and 0.97. These results indicate that the general published soils data are sufficient for simulating daily outflows at the watershed outlet at this scale of watershed modeling.

Model predictions of daily flow rates for agricultural subwatershed T4 (Fig. 2) were not as good as S4. Average absolute daily deviations of 1.76, 0.74, and 0.72 mm/day and correlation coefficients (r) of 0.80, 0.87, and 0.94 were obtained for years 1996, 1997, and 1998, respectively. There are several reasons for these large discrepancies. The overpredictions for days 262 and 282 were during tropical storms when accurate flow measurements were not available. Underprediction of the flows for the large event on day 600 (1997) was attributed to irrigation of the agricultural fields in weeks prior to the event. The irrigation was not taken into account in the model because the amounts and timings of irrigation were not known. The relatively dry antecedent conditions simulated by the model also caused the peak rates of events occurring in early 1998 (days 756-780) to be underpredicted. Some other discrepancies on days 420-470 and days 670-700 (1997) were due to ditch cleaning and removal of a beaver dam downstream in the outlet canal. Other potential problems in this agricultural subwatershed may be due to heterogeneties in soil types, cropping practices, water management practices (not taken into detailed account) in contrast with the more homogeneous forested subwatershed S4. On an annual basis the predictions of outflows were satisfactory for both S4 and T4. Assuming consistency in model structure, the results clearly demonstrate the need for detailed field information for making a more reliable prediction of daily outflows. This is especially true on lands with heterogeneous soils, crops and complex management practices.

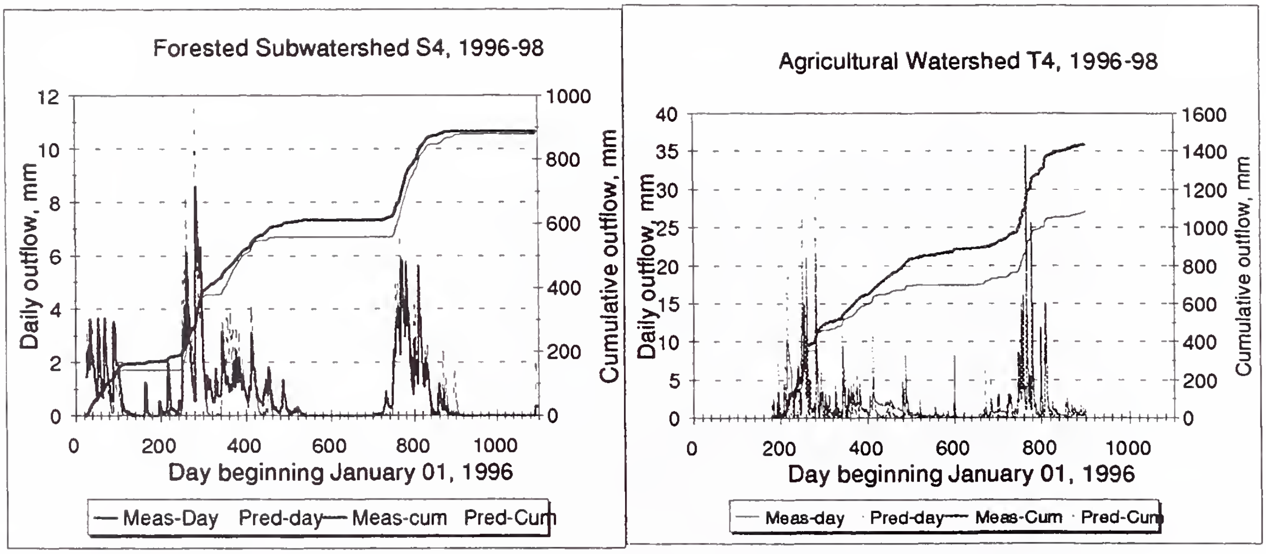
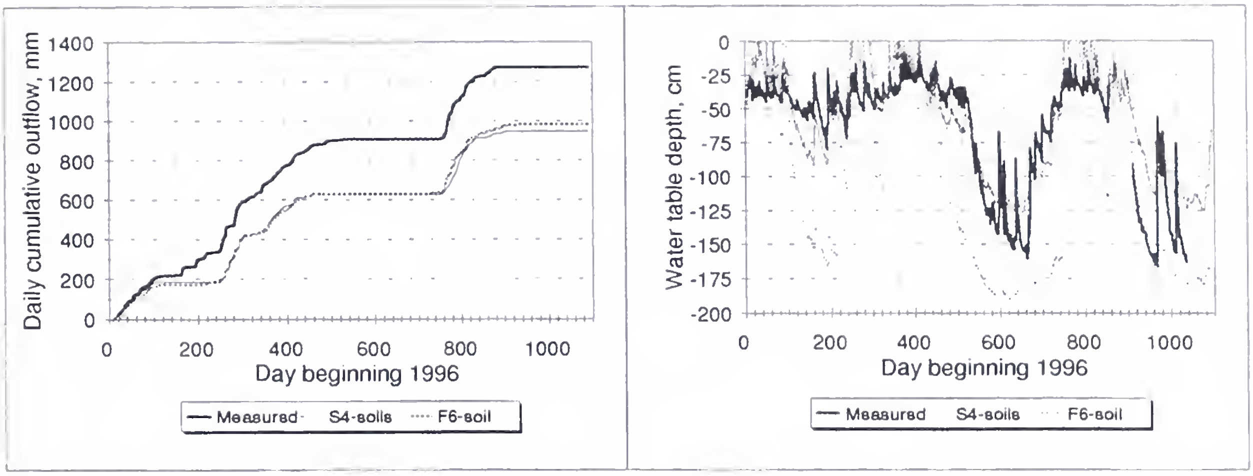


Figure 2. Measured and predicted daily outflows for the subwatersheds S4 (forest) and T4 (agricultural).

As discussed earlier, predictions of water quality parameters (nutrient loadings) at the watershed outlet and other in-stream locations depend upon the reliability of predictions of outflow rates and water table depths in the fields and in-stream hydraulics (velocities and water levels). A separate analysis was conducted to compare model predicted daily outflows at an in-stream canal outlet S2 (see Fig. 1) draining about 1460 ha of forests within the watershed. Peak

drainage rates were somewhat overpredicted (not shown) resulting in overprediction of total drainage by 25 % at the end of the 3-year period. Most of these overpredictions again occurred during large winter (1998) and summer tropical events (1996) that caused weir submergence not only at this outlet, but also almost at all field and in-stream locations. Otherwise, model predictions of all drainage events were in reasonable agreement with measured data. The computed



average daily absolute deviation and correlation coefficient (r) were 0.37 mm/day and 0.91, respectively, indicating the model's capability to make reasonable predictions of in-stream hydraulics (velocities and flow rates in this case).

Fig. 3. Measured and predicted daily cumulative outflows (left) and daily water table depths (right) for field F6 within the forest subwatershed S4. S4-soils is for original data and F6-soil is data measured at F6 field.

Daily field hydrology would be used by the mechanistic water quality model DRAINMOD-N to simulate water quality at the field edge. Daily outflows and water table depths predicted by the model (DRAINWAT) at the outlet of experimental field F6 (see Fig. 1) within the subwatershed S4 were compared with the measured data for that field (Fig. 3) to test the reliability of the model. Daily outflows were underpredicted during days 180-240 (summer 1996) and days 420-480 (spring 1997) resulting in total reduction of cumulative outflow by as much as by 26 % for the three-year (1996-98) period. This could be due to either overprediction of ET by the model for this field with 4-6 year old pine trees and/or errors in the soil hydraulic properties obtained from the literature and used in the model for this field. This was evident from the comparison of measured and predicted daily water table depths. The large discrepancies in water table depths throughout the simulation period (see S4-soils in Fig. 3) were most likely due to the large errors in estimated soil hydraulic properties and potential ET. Error due to variability in rainfall was ruled out because data was used from the gauge R6 in this field. Using some measured soil hydraulic data from that field (F6-soil curves in Fig. 3) in the model, predictions of water table depths were much improved. But the cumulative drainage was only slightly improved. These results indicate that the excellent performance of this distributed model in predicting daily drainage rates at the watershed outlet (S4), when general soils data were used, did not hold true for predictions at a location within the watershed. This was not surprising because the model was not calibrated at all with field measured data. The results are consistent with the discussions by Styczen and Storm (1993) in their study of nitrogen movements on a catchment scale. The study clearly supports the need of multi response data (other than outflows) to better identify model parameters and structure for an accurate validation of these models as was reported by Kuczera and Mroczkowski (1998). One such set of data for this type of model is water table depth from individual fields (Fig. 3). It is, therefore, clear that more effort is needed to determine inputs for this comprehensive model, if we are to use DRAINMOD-N for simulating daily water quality at the field edge. However, we hypothesize that, even with minimum calibration efforts, DRAINMOD-N linked with this watershed hydrologic model may still be applied, with lumped parameter in-stream water quality transport models, to predict seasonal and annual nutrient loadings from these poorly drained watersheds.

Summary and Conclusion

A DRAINMOD-based distributed watershed scale hydrologic/water quality model was tested using nearly three years

of outflow data from a 2950 ha forested watershed and a 710 ha agricultural watershed on relatively flat, poorly drained soils in eastern North Carolina. The model predictions of daily drainage rates were in much better agreement with measured data for the forested watershed than the agricultural watershed, which has more heterogeneous land management practices. Results showed that published soil property data can be used as a preliminary source for predictions of outflow rates from large watersheds at least during the planning phase. Model's reasonable prediction of in-stream flow rates (velocities and depths) within the watershed indicated its potential for linkage with in-stream transport water quality models that can be used to predict seasonal/annual cumulative water quality impacts on coastal watersheds. However, for an accurate prediction of hydrology on a smaller time scale, both at the watershed outlet and within the watershed, more effort is needed to determine input soil properties, or to calibrate the model. Multi response data such as water table depths and flow rates can be used to calibrate these models so that they can be used as comprehensive hydrologic/water quality models. Research is ongoing to further test the model at multiple sites for both flow rates and water table depths from multiple years to include the effects of year-to-year variation in weather. Tests of water quality predictions and nutrient loadings are also being conducted.

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Evaluating HSPF for Simulating Sediment Yield from a Claypan Agricultural Watershed in Central Missouri[1]

Abstract

The performance of Hydrological Simulation Program - FORTRAN (HSPF) version 11.0 in simulating sediment yield from a claypan agricultural watershed was evaluated. The calibration of model parameters required to describe sediment behavior constituted the major activity. The claypan watershed studied is Goodwater Creek, a 72 km^2 (28 mi^2) agricultural watershed in Central Missouri. Claypan soils are characterized by poor drainage resulting partially from the presence of an argillic claypan horizon that limits groundwater recharge and promotes surface runoff and interflow. The watershed is on a glacial till plain that has a level to gently rolling topography, with slopes averaging 3 percent. The low permeability of the claypan layer ensures that surface runoff is the dominant component of streamflow. The runoff water transports pesticides and nutrients from the watershed either in solution or adsorbed onto sediment particles. These agricultural chemicals delivered to the stream threaten the quality of the stream water. Thus, a clear understanding of the sediment process is essential for a successful evaluation of stream water quality as impacted by pesticide and nutrient loading.

Sediment problems in the Goodwater Creek watershed have not received any significant study. In this study the HSPF model was first calibrated for land surface sediment erosion on a 36 ha (89 ac) research field within the Goodwater Creek watershed. Calibration for instream sediment transport was done for the entire Goodwater Creek watershed. The calibration was based on two years (1993 and 1995) of observed sediment concentration data. The year 1994 was dry and not significant for sediment study.

Our results show that with a proper calibration of hydrology and sediment parameters, HSPF simulated sediment yield from the Goodwater Creek watershed accurately.

Keywords: HSPF, GIS, watershed modeling, water quality, runoff, and erosion.

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Introduction

The Hydrological Simulation Program - FORTRAN (HSPF) is a comprehensive, continuous model designed to simulate watershed hydrology and associated water quality. It considers a wide range of parameters that impact on hydrology and water quality. This raises the expectation that the scope of its applicability should be wide. Evaluation of the model that is essentially equivalent to calibration of its parameters is, therefore, an important effort toward understanding the model and bringing it into practical application. The application to claypan soils is of particular interest in this study.

Goodwater Creek watershed is located in the central claypan resource area of Central Missouri. It is a very important research watershed, in part, because the hydrologic and water quality characteristics of claypan soils are of considerable interest and not yet well understood. Efforts to gain knowledge of these characteristics would be significantly aided by a good water quantity and quality simulation model with the broad capabilities promised by HSPF model. Goodwater Creek watershed was established as a research catchment by the U.S. Department of Agriculture, Agricultural Research Service (USDA-ARS) in 1971, with these objectives: (1) to determine the influence of precipitation characteristics and watershed scale on water yield and flood characteristics; and (2) to determine the basic mechanics of the runoff and interflow process. Subsequently, the study of water quality was added as an objective in 1990.

Pesticides and nutrients are transported from the watershed either in solution in the runoff water or adsorbed onto eroded sediment particles. Therefore, accurate simulation of hydrology and sediment processes is a necessary requirement for a successful evaluation of pesticide and nutrient simulation results. Wang et al. (1999) have reported success in simulating the hydrology of Goodwater Creek claypan watershed using HSPF.

The objective of this study is to evaluate HSPF for simulating the sediment processes on the claypan watershed. Essentially, the model parameters that have significant bearing on erosion and sedimentation will be calibrated and model simulation of the sediment processes on a claypan watershed evaluated using historical data. Among the expected results of this exercise is the establishment of normal representative values of sediment parameters for claypan soils.

The HSPF Model

The Hydrological Simulation Program B FORTRAN (HSPF) is a complex model. It is, however, structurally modular, permitting the user to isolate and operate modules pertinent to solution of a given problem while bypassing other modules and their activities. Details of model structure are given by Bicknell et al. (1996) and are not treated here. Instead, this study gives, in a very broad manner, the model overview, while concentrating on activities which the user must perform to achieve a successful run. Thus, a great deal of time and effort was devoted to data preparation, model calibration, and evaluation.

HSPF considers a basin as comprising two segments: land segment and water body. The land segment is further disaggregated into pervious and impervious components. The water body, whether it be a stream reach or a lake/impoundment, is designated as a reach. The pervious land segment, impervious segment, and reach form three modules: PERLND, IMPLND, and RCHRES, respectively, of the HSPF model. The HSPF generates runoff and associated pollutants from pervious and impervious land segments and routes the flow down slope, over intervening land segments and into a stream reach, and then from one reach to another downstream to the outlet. A discussion of input data, with particular reference to parameters that require calibration for sediment simulation, is presented in a subsequent section under model parameters.

Study Watershed

Goodwater Creek watershed is a 72 km^2 (28 mi^2) agricultural watershed in Central Missouri, in the Central Claypan Land Resource Area. The watershed is on a glacial till plain that has a level to gently rolling topography, with slopes averaging 3 percent, but occasionally exceeding 5 percent (Jamison et al., 1968). The average longitudinal stream channel slope is about 0.36 percent. Predominant soils are the Udollic Ochraqualfs, Albaquic Hapludalfs, and Vertic Ochraqualfs of the Mexico, Adco, and Leonard series, respectively. The soil surface is covered with a layer of loess underlain by an argillic horizon at a depth of 15-30 cm below the soil surface. The clay content of the argillic horizon is generally higher than 50%. The low hydraulic conductivity characteristic of the claypan layer ensures that surface runoff supplies the only significant contribution to streamflow (Saxton and Whitaker, 1970; Hjelmfelt et al., 1999). The low

permeability of the claypan coupled with the relatively flat topography of the watershed causes wetness problems during the rainy season.

The main stream of Goodwater Creek is gaged at three locations, resulting in a nested watershed system with 18, 43, and 100 percent of watershed areas contributing flow to the three gages, respectively. Each gaging site is instrumented with a concrete v-notch weir, water stage recorder, and sampler to measure runoff and collect runoff samples for water quality analysis. Within the larger watershed, there are three research fields, designated as field 1, field 2, and field 3, whose outlets are also instrumented for flow quantity and quality measurements.

This study was conducted at the watershed and field scales. HSPF was calibrated for soil splash and sediment washoff using field 1. Calibration for sediment transport was done for the entire Goodwater Creek. Analysis and results are presented for the outlets of the larger watershed and for field 1.

Model Parameters.

The parameters required to execute an HSPF model run for sediment simulation can be grouped under the four categories: watershed, meteorological, hydrological, and sediment parameters. The first three have been discussed by Wang et al. (1999) and will not be treated here.

Sediment Parameters. The parameters that define sediment generation and transport comprise those parameters required to mathematically describe splash detachment of the soil matrix by rain drops, washoff of the detached sediment, scour of the soil matrix by overland flow, and sediment transport in the stream channel. The descriptive equations used to represent these processes in HSPF operation are given by Bicknell et al. (1996). The calibration process in this study is described below.

Initial Parameter Development. Bicknell et al. (1996) specifies the numerical range of HSPF model parameters. Donigian et al. (1978) provides general evaluation guidelines for setting numerical values of the parameters. Donigian et al. (1984) describes the entire application process of HSPF, in the Application Guide, to demonstrate the decisions, procedures, and results involved in a typical application. Donigian et al. (1977) presents a thorough discussion of the parameters in the application of the ARM Model to agricultural watersheds in Georgia and Michigan. Initial parameter values in this study were nominated by integrating information from the above sources with information from earlier HSPF-related experiences such as provided by Moore et al. (1988) and Chew et al. (1991).

Sediment Erosion Calibration. On pervious land segments, the major parameters that control the availability of sediment are JRER, KRER, JGER, and KGER. JRER and KRER represent, respectively, the exponent and coefficient in the soil splash equation of the sediment algorithm. JRER is related to rainfall intensity and incident energy to land surface. Based on the relationship between rainfall intensity and its incident energy proposed by Wischmeier and Smith (1958), several investigators have shown that soil splash is proportional to the square of the rainfall intensity (Meyer and Wischmeier, 1969; David and Beer, 1974). From these studies, a value of 2.0 is predicted for JRER. Donigian et al. (1984) observed that JRER is reasonably well defined and shows very moderate variability. Starting with JRER = 2.0, we arrived at a calibration value of 2.20. KRER, on the other hand, is related to detachability of soil type and surface conditions. It is related to the K-factor of the Universal Soil Loss Equation (USLE), and may be represented as K. The K-factor for a silt loam soil is given by Donigian and Davis (1978) as 0.43. Starting with this value, and noting that KRER has a lower limit of zero, we attained a calibrated value of KRER = 0.30. Sediment from gully erosion is affected by the coefficient (KGER) and exponent (JGER) in the matrix soil scour equation. Sediment from this source was considered negligible and was not accounted for in the calibration/simulation. Sediment washoff is controlled by JSER and KSER, the exponent and coefficient, respectively, in the sediment washoff equation. JSER has a narrow range (normally, 1.0 to 2.5). A value of 1.60 gave the best simulation result. KSER combines the effects of slope, overland flow length, sediment particle size, and surface roughness. With the present state of knowledge, KSER has a lower limit of zero but no known upper limit. Calibration is the major method of evaluating KSER. Starting from the calibrated value of KSER = 1.80 in the Iowa River study (Donigian et al., 1984), our adjusted value of KSER = 1.50 led to satisfactory simulation. DETS represents detached sediment storage on the land surface. Agents of detachment comprise mainly agricultural operations such as plowing, cultivation, harrowing, harvest, etc. It is represented as an initial value and is preset for a model run. It may be reset if it is expected that agricultural operations may give rise to production and exposure of soil fines in excess of the preset DETS value. The preset DETS in this study was 448kg/ha. To reflect field cultivations, DETS was reset to 5600 kg/ha to simulate sediment from field 1 for 1995. The reset value for field 1 for 1993 was 44 kg/ha. However, not resetting DETS gave the best sediment simulation for the entire Goodwater Creek for both 1993 and 1995.

Sediment Transport Calibration. HSPF accepts any of three methods for simulating instream sediment transport: Toffaleti method, Colby method, and user-specified power function method. In this study, the power function method was adopted. For the sand component of sediment, the coefficient (KSAND) and exponent (EXPSND) of the sandload power function formula were found, by trial, to have values of 25.0 and 2.0, respectively, for a good simulation. For the cohesive sediments (silt and clay), most of the input parameters were deterministically evaluated from experience, literature, or physical measurements. These include such parameters as particle size and density, and geometry of the transport channel. The calibrated value of erodibility of sediment (M = 1.80) from the Iowa study (Donigian et al., 1984) was adopted and initially kept constant. Thus, the parameters that required major consideration were reduced to two: critical bed shear stress for scour (TAUCS) and critical shear stress for deposition (TAUCD). The calibration process for cohesive sediments is illustrated with the calibration of critical shear stress for impending motion (TAUCS) of a silt particle. Information available in the literature suggests that the critical shear stress is highly variable, depending on among other things, the size and shape of the sediment particles themselves, particle orientation and angle of repose, bed roughness, water temperature. For this study it was decided to compute the shear stress for impending motion for average size sediment particle. For silt, Gottschalk (1964) gave a particle size range of 0.004 - 0.062 mm. A value of 0.016 mm was selected to represent average particle size. For the average particle size of 0.016 mm in water at 10° C, TAUCS was calculated as 2.637 g/m^2 by Wiberg and Smith (1987) method. TAUCD was selected as one tenth of TAUCS. The erodibility coefficient (M) was varied to improve the prediction as recommended by Donigian et al., (1984). A final calibrated value of 3.0 was attained.

The entire Goodwater Creek watershed, including field 1, was assumed to be totally rural in character so that the IMPLND module was not applicable. Thus, only the parameters required to operate PERLND and RCHRES modules for runoff and sediment simulation were calibrated.

Results and Discussion

The agreement between simulated and observed annual sediment was very good. The agreement between monthly sediment was fair to good. At the outlet of field1, the simulated annual sediment loss was 331.93 kg/ha for 1993 and 4058.66 kg/ha for 1995. The corresponding observed sediment losses were 262.69 kg/ha and 5609.55 kg/ha, respectively. At Goodwater Creek outlet, the simulated annual sediment losses were 5701.77 kg/ha and 1564.26 kg/ha for 1993 and 1995 respectively. The corresponding observed values were 4211.57 kg/ha and 1233.91 kg/ha, respectively. The coefficient of determination (R^2) was 0.99 for field 1 and 0.70 for Goodwater Creek. Differences between observed and simulated sediment may be attributed to several factors. DETS is difficult to calibrate or simulate because it is not possible to predict precisely how much soil fines would be exposed by any of the factors that yield loose soils. Also, the amount of soil fines that a particular operation exposes depends on many factors, including the soil moisture content. Another probable cause of discrepancy is that this study simulated sediment continuously while actual sampling was carried out at intervals. In some cases, only one sample was taken over a one-month period. Under this set-up, it should be expected that the simulated values should be higher than the observed. This was generally the case.

Interestingly, sediment at the outlet of Goodwater Creek was simulated accurately without resetting DETS. A probable explanation of this finding is that sediment washed off each of the 32 subwatersheds had to travel long distances over vegetated areas to reach the stream channel. There was ample opportunity for the sediment to be trapped and retained.

This study was based on two years of data. Obviously, a longer record would have been preferable, but that was not possible. However, twelve months of each year were simulated, giving a total of 24 simulations. A basis of confidence in the calibration/simulation is that one set of parameters was used to obtain good simulation for 24 months that embodied contrasting hydrometeorological conditions. Also, the parameters calibrated for field 1 were used without modifications to simulate sediment from the entire watershed. Chew et al. (1991) performed a similar study with a longer record but on a different soil and their findings were similar to those reported here.

Conclusions

Modeling field activities through resetting DETS is very difficult. This is because the variables that determine the magnitude of DETS are difficult to control. Serious over-estimation of sediment loss can easily result from incorrectly resetting DETS. From the result of this study it could be concluded that as the catchment area increases, the gains of resetting DETS diminishes significantly. This would be very fortunate in view of the difficuly in estimating correct DETS reset values.

It was concluded from the result of this study that HSPF can simulate sediment loss from a claypan soil very successfully.

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PART 2 - ADVANCES IN WATER QUALITY MODELING II

SESSION MODERATOR:

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Quantification and Effects of In-Stream Processes in the Ditches and Canals of the Lower Coastal Plain of North Carolina[1]

Abstract

Excess nutrient loads have been recognized to be the major cause of serious water quality problems recently encountered in the North Carolina estuaries and coastal waters. There has been a particular concern in coastal watersheds because agricultural and forested lands are located in close proximity to recreational and environmentally sensitive waters. In-stream modeling is an important tool for predicting water quality at the outlets of such watersheds. However, the value of these models is heavily dependent on the availability of in-stream rate parameters. A study has been initiated near Plymouth, NC to quantify in-stream processes in a lower coastal plain watershed. General nutrient mass balance components are being measured as well as rates of more specific individual processes. Nutrient mass balance was performed in an isolated 1,160m long agricultural canal section and in a collector canal draining a 84 ha ditched agricultural block. Flow and nutrient loads were measured at upstream and downstream stations on the isolated canal section and at the outlet of the collector canal using flumes and weirs equipped with velocity meters and automatic water samplers. Water quality was monitored on a continuous basis by sampling canal water on a flow-dependent frequency. Retention of nitrate in the 1,200m canal section varied from 15% in a 13-day period in June 1998 to 70% in a 33-day period in July and August 1998. Similar retention values were measured for total nitrogen, total phosphorus, and sediments. Nutrient concentrations monitored along the ditches and collector canal of the block showed a decrease in concentration as the water flows toward the outlet. Macrophyte biomass was measured in the canal section and the collector canal to quantify nutrient storage in plant tissues. Nutrient uptake kinetics for attached algae was also measured. Inorganic forms of nitrogen and phosphate in the sediment pore water of the canal section were measured on a weekly basis to quantify fluxes at the sediment-water interface.

Keywords: in-stream processes, water quality modeling, watershed scale

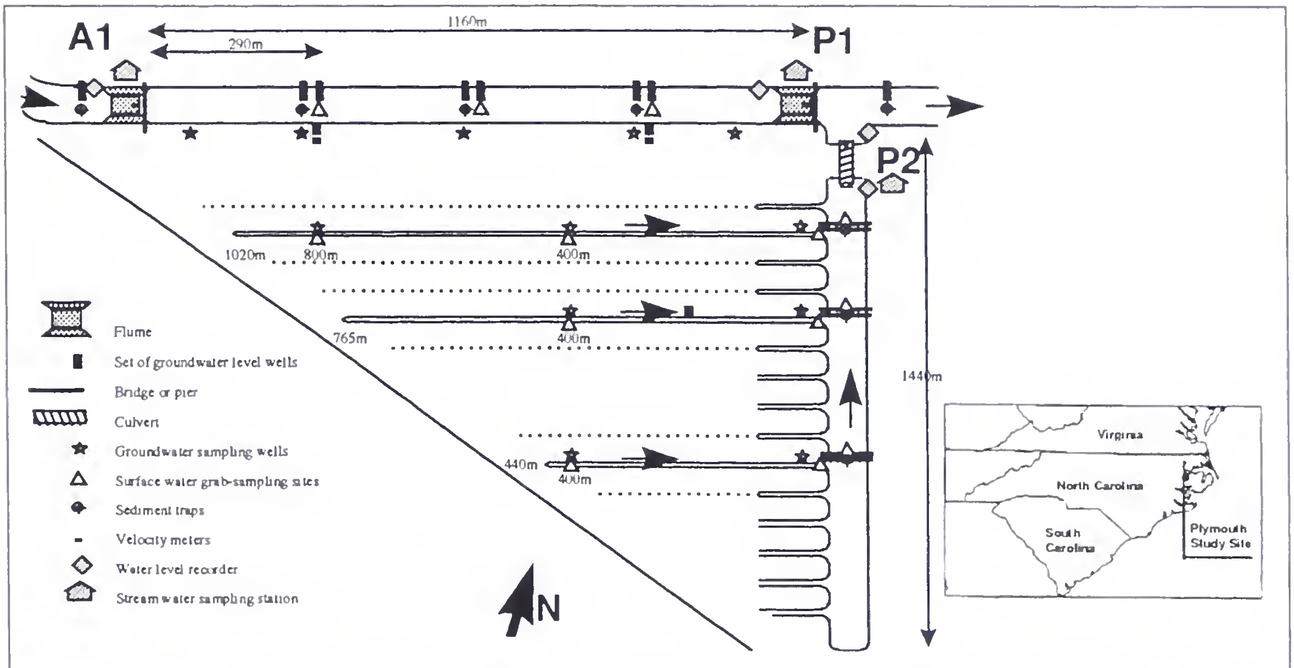

Figure 1 Schematic layout of the two sites where in-stream measurements are performed, near Plymouth, NC. The canal section is represented at the top between A1 and P1 and the agricultural block below it (not to scale).

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Introduction

Coastal regions in most developed countries are experiencing more and more serious environmental problems as the apparent increase of harmful algal blooms in coastal and estuarine waters suggests (Hallegraeff, 1993). North Carolina is certainly no exception to this trend. Serious algal blooms have been recorded as early as the early 1970s in the North Carolina Sounds and estuaries. Chronic environmental problems include: noxious algal blooms in the Chowan and Neuse Rivers; outbreaks of fish disease, increase of fish kill numbers, large sediment loads and low dissolved-oxygen concentrations in the Tar-Pamlico system; declines in submerged macrophyte populations in the Pamlico River; and declines in fish stocks and changes in salinity regimes throughout the Albemarle-Pamlico system (North Carolina Department of Environment, Health, and Natural Resources, 1993). Some fish kills have since been linked to the dinoflagellate alga *Pfiesteria piscicida* (Burkholder *et al.*, 1992).

Serious environmental problems have been linked in part to the general increase of nutrients and sediment loads in the estuarine systems. Nutrient loads have increased rapidly with population growth and waste input. Agriculture has been identified as a major source of nutrients entering the North Carolina estuarine systems (Danielson, 1994). There has been a particular concern in coastal watersheds because agricultural lands are located in close proximity to recreational and environmentally sensitive waters. Water quality research has primarily been conducted at the field scale and good information is already available to evaluate nutrient and sediment loading at the field edge for typical land uses encountered in the coastal plain of North Carolina. The missing link for understanding regional water quality impacts is quantification of changes that occur in drainage canals and small streams as the water moves from the fields to receiving rivers and estuaries. This information would allow integration of land uses and management practices to reduce pollutant loads and improve water quality in receiving waters.

Research in the last twenty years has shown the importance of in-stream processes to explain nutrient deliveries at watershed outlets. The general trend, reported throughout the literature, is a decrease of nutrient concentrations in streams as water flows downstream of a particular watershed (Meybeck, 1982 among others). Although some of this trend could be explained by a dilution effect in some cases, many authors have shown that nutrient retention does occur along canals, streams and rivers. Kuenzler (1991) showed that in a Coastal Plain stream receiving wastewater effluents, 60 to 100% of the nutrients were removed in distances varying from 1.26 to 3.75 km. Nitrate retention of 40 to 75% in streams during low flow periods are commonly reported (e.g. Hill, 1979,1988; Jansson et al., 1994).

Although trends are clear in the overall effects of in-stream processes, they remain very site specific. Since the 1970s, researchers have studied in-stream processes and have divided them into elemental reactions in an attempt to model their effects. Very complicated and powerful models have been written to take into account all processes (e. g. WASP, Ambrose et al., 1988). Although, those models were primarily designed to predict water quality in large water bodies such as lakes, rivers and estuaries, many of the processes considered occur in much smaller water systems such as ditches and canals. However, the accuracy relies almost entirely on the modeler's ability to use correct chemical reaction rates. Our hypothesis is that in-stream processes in the ditches and canals of the lower Coastal Plain of North Carolina can be described and quantified. This can be done in such a way that existing models can be adapted to the existing conditions in the Coastal Plain and that rates and constants can be measured directly in the field. The objectives of the research presented in this manuscript are to: (1) show the experimental setup used to measure in-stream processes in the ditches and canals of the lower Coastal Plain; and (2) to report preliminary results and observations.

Basic Hypothesis and Methods

The methods and experiments used in this study are adapted to the hydrologic functioning of the region. The lower coastal plain of North Carolina is naturally low, flat and wet. A dense drainage network (drainage density > 15 Mi/mi^2) was constructed, mostly between the 1950s and the 1970s to provide sufficient drainage for agricultural and forestry production. The normal hydrology of the system may be described as follows: during a rain event, the water table rises in the soil. For large rainfall events the water table may rise to the surface and surface runoff to the drainage ditches may occur. But for most events, drainage is by subsurface flow to the lateral ditches, and to a lesser extent, directly to collector and main canals. Water in the drainage ditches then flows into collector canals and then to receiving main canals. Canal water usually empties into natural streams before it reaches the nearby estuaries and sounds. Our hypothesis is that nutrients in the water are submitted to in-stream processes from when they enter the ditches until they reach the estuaries. The study reported here focuses on the processes occurring in agricultural ditches and canals in the coastal plain.

One of the main objectives of this study is to provide field-measured data to serve as input in water quality models being developed for coastal watersheds. Quantifying in-stream processes therefore implies measuring individual process rates that can be used as input data in the models. Because the aim of the modeling approach is to predict overall changes in the ditches and canals as water flows downstream, it is desirable to directly measure those effects in the field. Thus, our approach to quantifying in-stream processes has two complementary components: (1) describe and quantify each individual process and determine values and ranges for each process rate and, (2) quantify the overall changes of in-stream processes on isolated reaches and fields by conducting a nutrient mass balance.

These two approaches are being conducted on two separate sites located in the coastal plain near Plymouth, NC. The first site focuses on processes in ditches and a collector canal in an agricultural block and the second site focuses on processes occurring in an isolated main canal section. The two sites happen to be located next to each other (see Figure1). This is a convenience so far as conducting the field work is concerned, but not necessary to satisfy the objectives of our research.

Equipment Installation and Setup

Canal Section - A 1,160m long canal reach has been isolated and instrumented (see Figure1). Its primary purpose is to determine a nutrient mass balance and thus measure the overall effect of in-stream processes on nutrients in agricultural canals. Water flowing through the section drains from a mixed agricultural and forested 1,640 Ha watershed. The canal averages 1.4m in depth, 10m top width and 4.5m bottom width over the length of the reach and is well dimensioned for the flow it must carry. Water enters the section at station A1 and leaves it at station P1. Flow and water quality parameters are monitored continuously (details later) allowing computation of nutrient fluxes at both A1 and P1. The difference between the two fluxes is assumed to be the net result of the nutrient transformations in the section.

The canal reach is not a totally isolated system. Inevitable addition of groundwater and nutrients occurs from the adjacent agricultural field to the south and from the forest to the north. Water table profile in the field is measured continuously at two locations along the reach to provide information on the seepage rate into the canal while nutrient concentrations in the groundwater are sampled on a weekly basis at five locations along the reach (Figure1). Unexpected water from a forested block to the north seeps under the forest road into the study reach. Seepage rate is also estimated and nutrient concentrations are measured on a weekly basis.

The reach has also been equipped for the measurements of individual processes. Wooden piers have been installed over the canal at three intermediate stations between A1 and P1 to provide easy access over the canal without disturbing the banks or the bottom sediments. Sediment traps have been installed near the piers to measure gross sedimentation (Figure1).

Agricultural Block - The 84 Ha triangular-shaped agricultural block has been instrumented primarily to measure in-stream processes in the ditches and collector canals. Unlike the case of the canal section, there is no obvious way to perform a nutrient mass balance in the ditches because the water inputs are the surface and subsurface drainage and there is no practical way to continuously measure fluxes of water and nutrients entering the 15 ditches (Figure1). Our hypothesis is that, if nutrient transformations in the ditches and collector canals are quantitatively important, they should be measurable along the ditches and the receiving canal. Three ditches in the block were chosen and instrumented. Groundwater wells within 1m of the ditches were installed 400m apart (Figure1) to measure the nutrient concentration in the groundwater just before it enters the lateral ditches. Groundwater and ditch water are sampled on a biweekly basis, or more often when necessary, every 400 m along the ditches. In addition, water is sampled along the collector canal at the same frequency. Flow and water quality parameters are measured on a continuous basis at station P2 at the outlet of the block (Figure1). The continuous recording of flow and water quality leaving the block may sound like a luxury, because the observable effects of in-stream processes are actually provided by the biweekly spatial sampling. However, valuable detailed information on how processes occur in time is obtained this way.

Flow and Water Quality Measurements

The value of the nutrient mass balance study in the canal section depends on the accuracy of the nutrient flux measurements at both A1 and P1. The actual instantaneous nutrient flux at one station can, ideally, be calculated at any particular time by multiplying the flow by the nutrient concentration. The total nutrient load going through one station is thus the integration over time of all instantaneous fluxes. Our hypothesis is that we can take

measurements frequently enough to be able to recreate a flow and concentration time series that closely approximate the actual one.

Storm runoff characteristics from small and large watersheds are generally different. Response time is short, and runoff depth and unit area peak runoff rate are higher for small watersheds than for larger ones (Garbrecht, 1991). Preliminary data showed that flow could increase several fold within hours and sometimes minutes at stations A1 and P2, which was expected given the small size of the watersheds. We hypothesized that any dramatic hydrologic response should be accompanied by a corresponding change in nutrient concentrations in water. Most hydrographs having fast rising and slower falling limbs: measurements are taken at a higher frequency during the rise than during the fall.

Two methods are used to estimate flow. The common weir setup is used to measure flow at the outlet of the agricultural block. Water in the collector canal is forced to flow over a set of two V-notch weirs before it can leave the block. Water elevation up- and downstream of the weir is recorded continuously and flow values are calculated using known relationship between water elevation above the invert of the weir and the flow. No similar methods could be used in the main canal because of the very low gradient and the amount of the water the canal must carry at times. A flume design is used instead. At both A1 and P1 stations, water is channeled through a trapezoidal-shaped flume of known dimensions. The velocity of the water is measured at the middle of the flume using STARFLOW Ultrasonic Doppler instrument from UNIDATA Australia. This velocity meter, placed at the bottom of the flume, is very well suited for the conditions in the canal section because it measures particle velocities up to one meter in front and above the sensor; this distance that allows the ultrasonic beam to scan the whole water column in most cases. Estimated velocities thus correspond to the average velocity of the water column above the sensor. The sensor takes a reading every 20 seconds and records an average of the readings every 10 minutes. A relationship between the velocity at the center of the flume and the average velocity for the whole section has been established by measuring velocity profiles within the cross section. STARFLOW also records the elevation of the water above the sensor from which the cross section area is calculated. The flow at A1 and P1 is finally estimated by multiplying the average velocity by the cross section area. Thus a flow value for both stations is calculated every 10 minutes regardless of the rise or fall of the hydrograph.

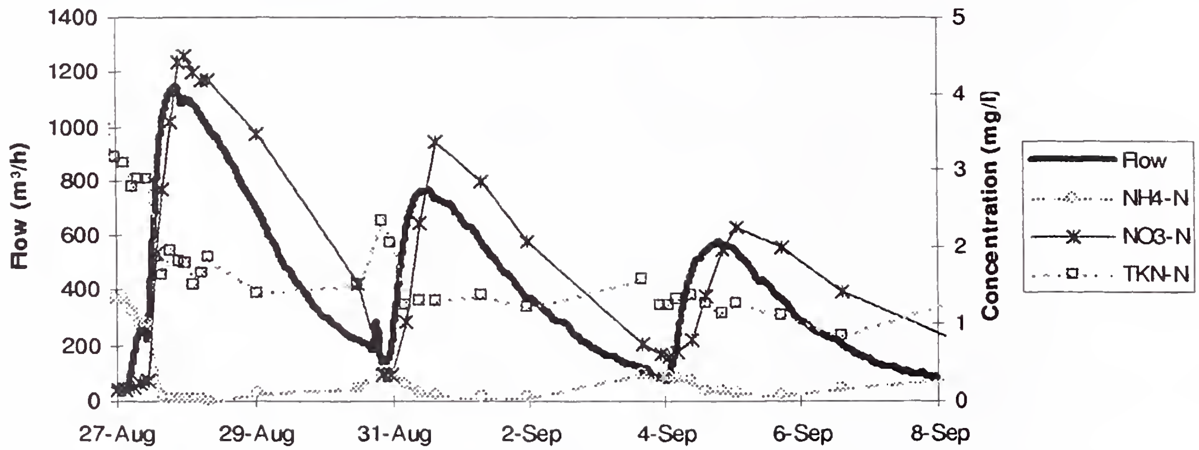

Figure 2. Flow and nitrogen concentration time series at station P1 in August and September 1998

Nutrient concentrations are obtained by collecting discrete water samples, which are later analyzed in the laboratory. One-liter water samples are drawn automatically using SIGMA™ 900 automatic sampler. The signal to sample is sent by a BLUE EARTH™ microprocessor that recognizes when the water level is rising or falling at the stations. To do so, the microprocessor is linked to a water level measuring device. A float follows the water level movement freely inside a pipe protecting it from external influences such as wind, floating debris and animals. It is connected to a counterweight by a pulley, which in turn is connected to a potentiometer. The resistance created at the potentiometer thus varies with the water level and is read with the microprocessor. A simple relation between the potentiometer reading and the water level is then used to derive the water level, which in turn is read and kept in live memory. A small program in the microprocessor then decides if the water level has changed significantly since the last reading and if the water level is rising or falling. According to prescribed criteria the program decides if the reading should be recorded and decides on the water sampling frequency. During the rising limb of a hydrograph, water is sampled every 2 hours and 40 minutes at A1 and P1 and every 3

hours at P2. During the falling limb, water is sampled every 16 hours or after an estimated volume of water has gone through the stations. This decision-making device based on water level is also used at P2 to calculate the flow according to the method discussed in the previous paragraph.

For each water quality parameter analyzed, a set of discrete concentrations, varying with time, is obtained. We hypothesize that the sampling and analysis of the samples are frequent enough so that a linear interpolation between two discrete values of concentrations is a good approximation to the actual concentrations in the field. By doing so, a concentration value can be associated with every recorded flow value. This way an instantaneous flux for all water quality parameters is estimated every 10 minutes. Nutrient load going through the stations is thus estimated by integrating over time those estimated fluxes. Figure 2 illustrates the flow and concentration time series obtained in August and September 1998 at P1.

Water samples are collected from the automatic samplers on a weekly basis or more often during rainy periods. A preliminary study showed that nutrient decay in the samples is not noticeable until the temperature in the sampler rises significantly in the summer times. All samples (automatic, grab and well samples) are kept on ice upon collection. They are frozen upon arrival in the laboratory and then stored at 4°C until analysis. Nutrients analyzed include Nitrate (NO_3-N), Ammonia (NH_4-N), Total Kjeldahl Nitrogen (TKN-N), Phosphate (PO_4-P), Total Phosphorus (TP-P), Total Dissolved Carbon (TDC) and Dissolved Organic Carbon (DOC). Total Suspended Solids (TSS), Chloride (Cl^-) and pH are also measured on each sample.

Individual Processes Measurements

In-stream processes involve many physical, chemical and biological reactions. It would be futile to try to estimate the rates of all reactions involved because of their number. One must choose, among the total, the ones that have the most effect on the water quality parameters of interest for a particular study. Existing in-stream models are very helpful in deciding the important and sensitive processes to be measured. It is our hypothesis that most of the nutrient transformations in ditches and canals occur at the sediment-water interface and, to a lesser extent, at the plant-water interface. Not all process rates have been measured at this point but more experiments are being set up. Macrophyte and filamentous algae biomass have been estimated in the canal section and the collector canal to quantify nutrient storage in plant tissues. Nutrient uptake rates for attached algae have been estimated. Filamentous algae were put into a clear Plexiglas tank and the decrease of nutrient concentrations in the tank were recorded with time.

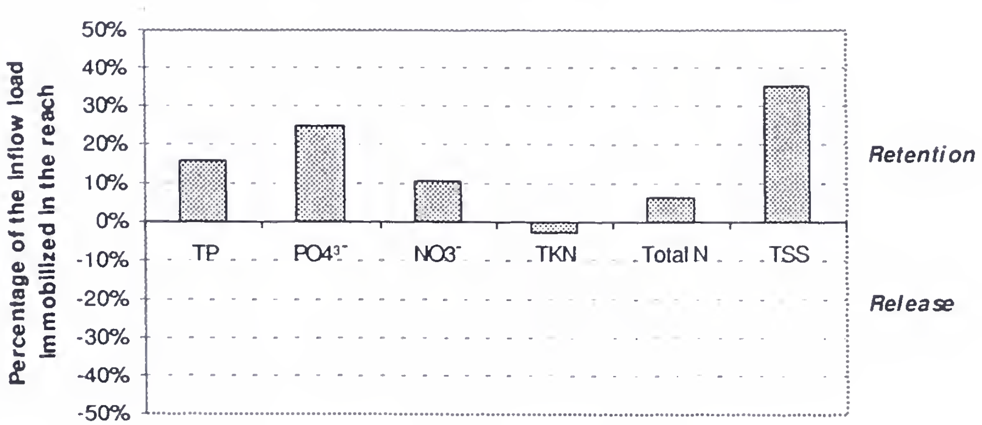

Figure 3. Retention of nutrients and Total Suspended Solids in the canal reach during a period of 15 days in June 1998

Ammonia flux from the sediment into the water column is estimated on a weekly basis. Concentration of inorganic forms of nitrogen and phosphate in the sediment pore water is measured using the pore water sampler described by Hesslein (1976). Ammonia, nitrate and Phosphate concentration profiles in the sediment and the water column are obtained on a weekly basis. Nutrient fluxes from and to the sediment are approximated from such profiles. Nitrification and denitrification rates in the water column and at the sediment-water interface are still to be estimated at this point.

Preliminary Results and Discussion

The general trend observed after preliminary analysis for data obtained in summer 1998 is a nutrient retention in the ditches and the canal reach. The nutrient concentrations along the ditches and the collector canal in the agricultural block consistently decrease as water moves downstream. More analysis of the data is necessary to draw firm conclusions at this point. Early results reported by Toulouse (1998) show that during two periods in summer 1998, nutrients and TSS were retained in the canal reach (Figure 3). During a 33-day period in July and August 1998, as much as 70%, 45 % and 55% of the loads of nitrate, Total Phosphorus and TSS respectively, entering the reach were retained. These values are rather high but are within the range of reported literature values obtained during low flow conditions (Kuenzler, 1991, Jansson et al., 1994). Such high percentages of nutrient retention can be interpreted as the combination of several factors. Chemical and biological processes are enhanced by warm temperatures. Although low flows do not occur only during warm periods, values reported here were estimated during summer. Because of the small volume, water has little momentum and its movement is easily slowed by debris and aquatic vegetation. As a consequence, there is more contact between processing sites and nutrients and, the nutrients are exposed to processes for longer periods of time. In addition, the absolute amount of nutrients flowing through the reach being relatively small, the proportion to be processed is likely to be higher for a constant processing rate. These factors taken separately or their combination make those percentages likely to be high. Similar numbers should not be expected for high flow conditions since the conditions for the factors described earlier can be opposite, especially in winter. The study is being pursued over a full hydrologic cycle. Valuable information on nutrient retention on event, monthly and annual basis will be estimated.

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Procedure to Estimate Soil Erodibility for Water Management Purposes.[1]

Abstract

One of the geotechnical aspects that controls the morphology of the landscape and performance of earthen structures experiencing concentrated flow is the erodibility of the earthen material on the land surface. Prediction of the erodibility of soil material by flowing water is complex. A submerged jet-testing device has been developed for characterizing soil erodibility in the laboratory and the field (Fig. A1). This device has been developed based on knowledge of the hydraulic characteristics of a submerged jet and the characteristics of soil material erodibility. The test is simple, quick, and relatively inexpensive to perform. The test is repeatable and gives consistent results, and the coefficients obtained from test results can be used in current equations to predict erosion.

The excess shear stress equation is commonly used to model the erosion of soil materials. Analytical procedures have been developed and are described for determining the excess shear stress parameters, coefficient of erodibility and critical stress, from the submerged jet test results.

This article contains a description of the method and procedures for jet testing in the field. An example is given of a series of tests comparing laboratory jet results, open channel test results (Fig. A2), and in situ jet test results (Fig. A3). The laboratory results indicate compaction water contents greater than 13% result in increased resistance within the tested range of water contents. The soil in the flume was compacted in the range of optimum water contents determined from the laboratory tests. This example verifies the utility of in situ jet tests to determine erosion resistance of a soil.

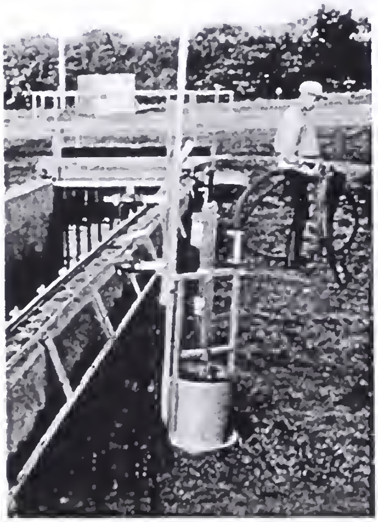

Figure A1. Jet test apparatus.

Figure A2. Open channel tests.

Figure A3. In situ jet test.

Keywords: jet testing, erodibility coefficient, critical stress, open channel flow, erosion

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Introduction

There are a number of water management problems related to the erosion of cohesive soils. These problems include river channel degradation and bank stability, and erosion associated with bridges, culverts, earthen spillways, road embankments, levees, and earthen dams. Prediction of the erosion resistance of earthen materials is integral to understanding the processes involved in concentrated flow erosion. The objective of this paper is to describe the apparatus and methodology for field jet testing. This includes the information to be recorded and steps of analysis. An example is also given of preliminary jet tests conducted in the laboratory to determine optimum placement conditions in the field and follow up confirmation of two open channel tests in a flume and in situ jet testing. The example indicates the utility of a jet test for determining soil erodibility for water management purposes.

Current Practice

It is common within most erosion studies to assume that rate of erosion ε is proportional to the effective shear stress in excess of the critical shear stress, and is expressed as:

$$\varepsilon = k_d(\tau_e - \tau_c) \tag{1}$$

where k_d is the erodibility coefficient, τ_e is the effective stress stress, and τ_c is the critical stress. Physically this equation indicates that in order to initiate erosion, the effective shear stress must be greater than the critical shear stress. The challenge is in determining k_d and τ_c for the soil material of interest.

The most dependable erosion test is a large open channel flow test with the material of interest forming the entire bed. This testing procedure poses many problems, particularly if the material to be tested is a native stream bed material. It is impossible to move that bed to a large open channel flume and test it over a wide range of stresses to observe its resistance without introducing a disturbance. Even for materials that are to be disturbed and remolded through compaction for construction purposes, it is difficult to justify conducting a large open channel test. Therefore there is a need for a method of testing these materials in the laboratory as well as in situ. A number of studies have used a submerged jet for testing of materials in the laboratory (Moore and Masch 1962, Hollick 1976, Hanson and Robinson 1993). A submerged jet test has been developed for testing materials in situ (Hanson 1991, Allen et al. 1997). Hanson (1991) developed a soil-dependent jet index that is based on the change in maximum scour depth caused by an impinging jet versus time. The jet index is empirically related to the soil erodibility. Testing apparatus and method for determining the jet index is described in ASTM Standard D5852-95.

In an attempt to remove empiricism and get direct measurements of the excess stress parameters τ_c and k_d, Hanson and Cook (1997) developed analytical procedures for determining the soil erodibility based on the diffusion principles of a submerged circular jet and scour beneath an impinging jet. These procedures are based on analytical procedures developed by Stein and Nett (1997) for a planar jet at an overfall. They validated this approach in the laboratory using six different soil types.

Stein and Nett (1997) showed that as the scour hole increases with time the applied shear stress decreases, due to increasing dissipation of jet energy within the plunge pool. Detachment rate is initially high and asymptotically approaches zero as shear stress approaches the critical shear stress of the bed material. The depth of scour at the point where the hydraulic shear is equivalent to the critical shear stress is called the equilibrium depth, H_e (Fig. 1). The critical stress for a circular submerged jet is determined by (Hanson and Cook 1997):

$$\tau_c = \tau_o \left(\frac{H_p}{H_e} \right)^2 \tag{2}$$

where H_p is the potential core length from the origin of the jet, H_e is the distance from the jet nozzle to the equilibrium depth of scour, and τ_o is the maximum applied bed shear stress within the potential core. The difficulty in determining equilibrium depth is that the length of time required to reach equilibrium can be large. Blaisdell et al. (1981) observed during studies on pipe outlets that scour in cohesionless sands continued to progress even after 14 months. They developed a hyperbolic function to compute

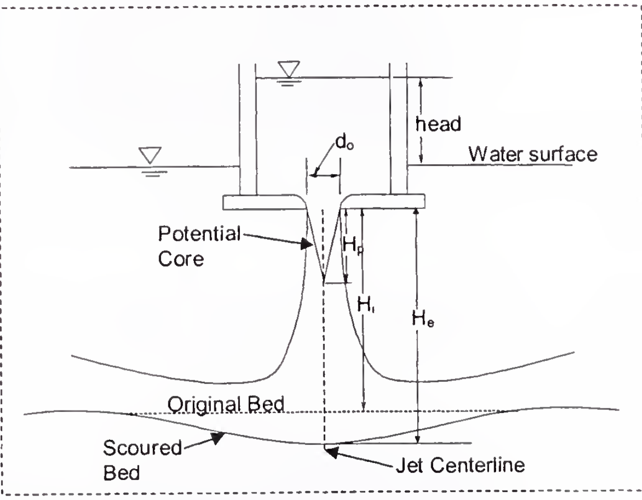


Figure 1 Schematic of jet scour parameters.

the equilibrium scour depth. This method assumes that the scour versus time follows a logarithmic hyperbolic function:

$$x = \left[(f - f_o)^2 - A^2\right]^{1/2} \tag{3}$$

$$x = \log\left(\frac{U_o t}{d_o}\right) \tag{4}$$

$$f = \log\left(\frac{H}{d_o}\right) - \log\left(\frac{U_o t}{d_o}\right) \tag{5}$$

where A is the value for the semitransverse and semiconjugate axis of the hyperbola, f_o is the asymptotic value of the hyperbola ($f_o = \log [H_e/d_o]$), U_o is the velocity of the jet at the origin, H is the distance from the jet nozzle to the maximum depth of scour at time t, and d_o is the diameter of the jet nozzle. Since x and f are known from Eq. 4 and 5, an iterative process is conducted in which the standard error is minimized based on the best fit values of f_o and A. The value for the equilibrium depth H_e is then determined based on the antilog of f_o.

The definition of the potential core length H_p and the maximum applied shear stress within the potential core τ_o is dependent on the diffusion properties of the jet. The potential core length is determined by the distance the centerline velocity of the jet remains equal to the velocity at the jet origin, U_o. The basic equation for modeling the erosion of a soil material beneath an impinging circular jet with an initial nozzle height H greater than the potential core length is (Hanson and Cook 1997):

$$\frac{dH}{dt} = k\left[\tau_o\left(\frac{H_p}{H}\right)^2 - \tau_c\right] \tag{6}$$

where dH/dt is the rate of scour. The maximum applied shear stress within the potential core is defined as $\tau_o = C_f \rho U_o^2$ where C_f is the coefficient of friction, and ρ is the density of water. This relationship is similar to the excess stress equation defined by Eq. 1. As scour occurs beneath an impinging jet the stress at the boundary and the rate of scour change. Therefore, an integrated dimensionless form of this equation (Hanson and Cook 1997) is used to analyze jet test results. The equation for measured time t_m is then expressed as:

$$t_m = T_r\left[0.5\ln\left(\frac{1+H^*}{1-H^*}\right) - H^* - 0.5\ln\left(\frac{1+H_i^*}{1-H_i^*}\right) + H_i^*\right] \tag{7}$$

where t_m is the measured time and $T_r = H_e/(k_d \tau_c)$ and dimensionless time $T^* = t/T_r$. The other two terms are dimensionless distance terms in which $H^* = H/H_e$ and $H_i^* = H_i/H_p$. The excess stress parameter τ_c can be predetermined by fitting the scour data to the logarithmic hyperbolic method described in Eq. 3, 4, and 5. The erodibility coefficient k_d is then determined by curve fitting measured values of H versus t for Eq. 7 and minimizing the error of the measured time t_m and the predicted time t_m.

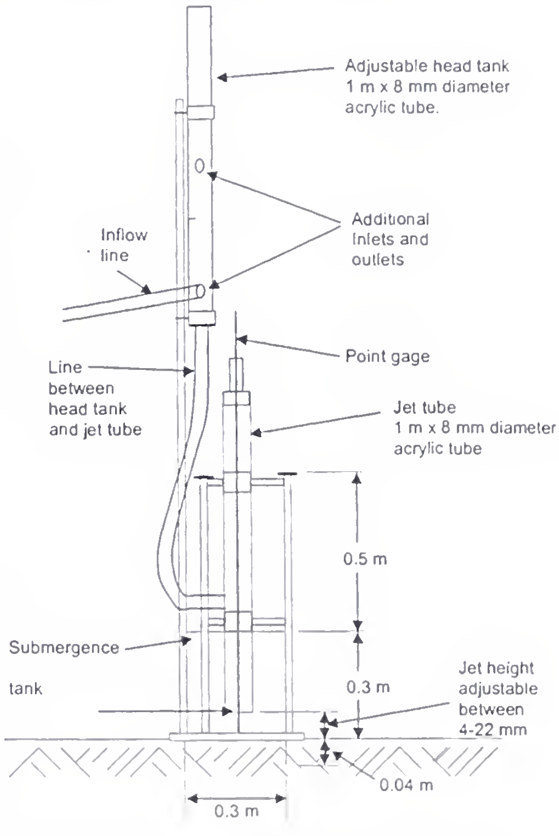


Figure 2. Schematic of in-situ jet apparatus.

Apparatus and Methodology for In situ Jet Testing

The in situ jet test apparatus consists of a pump, adjustable head tank, jet submergence tank, jet nozzle, delivery tube, and point gage (Fig. 2). Water is pumped directly from the channel bed into an adjustable head tank to allow the user to set a desired initial stress on the bed. The jet velocity at the nozzle origin U_o, jet height H_i, and jet diameter d_o, are the parameters adjusted to control the initial stress at the bed. With this apparatus an initial stress from 2 Pa to 200 Pa can be applied to the bed. The apparatus nozzle diameter is 6.4 mm. Once a test is started scour readings are

taken with a point gage. The point gage is aligned with the jet nozzle so that the point gage can pass through the nozzle to the bed to read the depth of scour. The point gage diameter is nominally equivalent to the nozzle diameter so that when the point gage rod passes through the nozzle opening, flow is effectively shut off.

Initiation of a jet test requires placement of the submergence tank by driving it into the bed 0.04 m. Once the submergence tank is set in the bed the jet tube and adjustable head tank is attached. The jet tube is attached to a hanger that orients the tube in the center of the submergence tank. The head tank is attached to a mast that allows the user to move it up or down. The head tank controls the pressure delivered to the jet nozzle and, in turn, the jet velocity. The operator, using markings on the side of the jet tube, estimates the initial height of the jet nozzle. Once the jet nozzle is set the initial jet height H_i is measured more precisely using the point gage. Following a determination of the bed elevation relative to the jet nozzle, the head is set by holding a plate downstream of the jet nozzle to divert the impinging jet. Once the head is set the testing can begin by removing the plate and allowing the jet to impinge on the bed. The time interval for measuring the bed surface is set by the operator.

A computer spreadsheet routine allows direct entry of the data and aids the operator in analyzing the scour data (Fig. 3). The essential data of a jet test includes the initial nozzle height and diameter, head setting, point gage readings, and time. The spreadsheet calculates the equilibrium depth of scour, critical stress, and the erodibility coefficient.

JET DATA

DATE 10/9/97

JET TEST LOCATION Station 53 in flume — OPERATOR gh

ZERO POINT GAGE READING 1.222 — TEST # 2

PRELIMINARY HEAD SETTING 87 — PT GAGE RDG @ NOZZLE 1.263

NOZZLE DIAMETER (IN) 0.2505 — NOZZLE HEIGHT (FT) 0.200

SCOUR DEPTH READINGS

TIME (MIN)	DIFF TIME (MIN)	PT GAGE READING (FT)	MAXIMUM DEPTH OF SCOUR (FT)
0	0	1.063	0.000
10	10	1.032	0.031
20	10	1.023	0.040
30	10	1.014	0.049
40	10	0.999	0.064
50	10	0.990	0.073
60	10	0.977	0.086
70	10	0.974	0.089
80	10	0.973	0.090

HEAD SETTING

TIME (MIN)	HEAD (IN)
0	87.00
10	87.00
20	87.00
30	87.00
40	87.00
50	87.00
60	87.00
70	87.00
80	87.00

Figure 3. Example data sheet.

The first calculation is the equilibrium scour depth. The hyperbolic function developed by Blaisdell et al. (1981) is used to compute the equilibrium scour depth based on Eq. 3 (Fig. 4). The τ_c is therefore estimated based on the equilibrium depth (Eq. 2). Once the equilibrium depth and τ_c are determined, k_d is determined from the measurements of scour depth and time (Fig. 5). The best value of k_d is assumed to be that value for which the standard error is a minimum for the predicted time versus the measured time of scour based on Eq. 7.

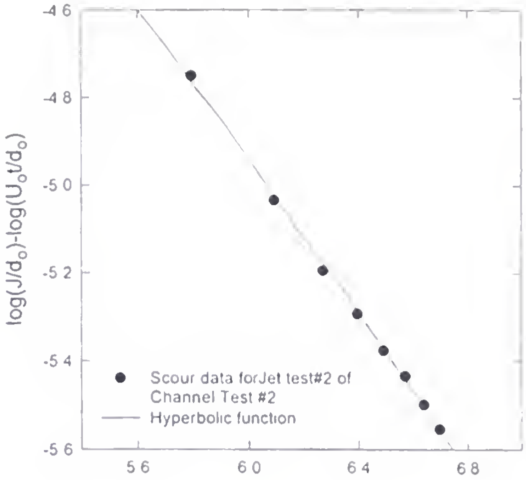
Example of Laboratory and Field Results

To reinforce the application of the jet test procedure and apparatus, an example of an actual testing process is presented. The example includes preliminary testing in the laboratory to determine optimum placement conditions of a soil for erosion resistance. Two open channel flow tests are conducted on the same soil compacted in the range of optimum water contents determined from the laboratory results. In situ jet tests are conducted on the placed soil and the results are compared to both the laboratory and the flume test results. The soil used in this example classifies as a clay loam (USDA texture) or CL (ASTM D2487). The gradation is 38% sand (>0.075 mm), 34% silt (>0.002 mm), and 28% clay < 0.002 mm), and the plasticity index (PI) is 15, and liquid limit is 26.

Laboratory Jet Tests

Laboratory jet tests were conducted on soil samples compacted by dynamic compaction at various water contents. Dynamic compaction was attained by dropping a 79.4-kg hammer from a height of 0.30 m. The soil was compacted in three layers using 24 blows per layer to attain a compaction effort of 603 kN-m/m^3. The compaction effort of ASTM Standard D698 is comparable at 600 kN-m/m^3. The modified compaction setup was used because larger samples than used in the ASTM Standard 698 were required to conduct the laboratory jet tests. The water content, wc%, was determined by the weight of water divided by the weight of solid in the soil element multiplied by 100. The dry unit weight, γ_d, was determined by the weight of solid in the soil element divided by the total volume occupied by the entire element. The dry unit weight and water content relationship for this soil is shown in (Fig. 6a).

The maximum depth of scour versus time for this soil at various water contents and dry unit weights was also determined for each sample. There is a dramatic decrease in scour observed with increases in compaction water content. A comparison of the water content and dry unit weight relationship (Fig. 6a) to the erodibility k_d and critical stress τ_c (Fig. 6b) provides insight into the potential benefits of proper material preparation and compaction. Compaction of the soil material at water contents greater than 13% result in maximum erosion resistance within the tested range of water contents.



Open Channel and In-situ Jet Test Results

Two open channel erosion tests were conducted on the same soil in a flume 1.8 m wide by 29 m long with 2.4-m sidewalls. A flat-bottomed channel bed 1.8 m wide and 21 m long was constructed in the flume for each test. Soil was placed in the flume on a 1% slope for test 1 and a 3% slope for test 2. The average water content of the placed soil in test 1 was 15.1% and in test 2 was 13.9%. Soil was placed in 15-cm loose lifts and compacted with 4 passes of a vibratory roller compactor, 2 passes without vibration and 2 with vibration. The resulting average dry unit weight was 1.80 g/cm^3 for test 1 and 1.85 g/cm^3 for test 2.

Flows were introduced in the flume, and water surface and bed surface readings were taken along the centerline of the channel for a 6-m test section to determine hydraulic stresses and erosion. For test 1 the discharge was set at 0.39, 0.79 and 1.58 (m^3/s)/m for time periods of 26, 17, and 11 hr, respectively. For test 2 the discharge was set at 0.38 and 1.58 (m^3/s)/m for time periods of 19 and 8 hr, respectively.

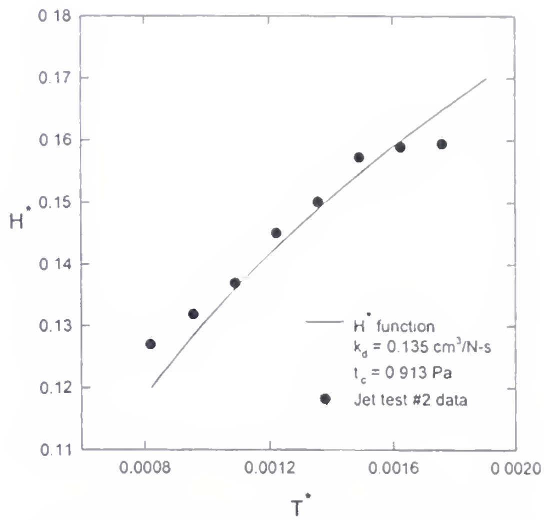


Figure 5. Dimensionless time function of scour to determine k_d.

In order to analyze the erosion data from the open channel testing it is necessary to use the same erosion rate model over the tested range of stresses as used in the jet testing (Eq. 1). Temple (1985) and Hanson (1990) utilized an integrated cumulative form of Eq. 1 to determine the erodibility coefficient k_d from vegetal channel and bare channel testing:

$$E_o - E_N = k_d \sum_{i=1}^{N} (\tau_e - \tau_c)\Delta t_i \tag{8}$$

where E_o is the bed elevation prior to flow, E_N is the bed elevation at time t, and Δt_i is the time interval for the ith flow depth measurement. The amount of erosion and average effective stress is determined from the data collected. The effective stresses varied from 12 Pa to 55 Pa over the range of testing in test 1 and 2. The determination of τ_c for this material based on Agricultural Handbook 667 (Temple et al. 1987) was 2.8 Pa for test 1 and 2.9 Pa for test 2. The value of k_d based on Eq. 8 was determined to be 0.06 cm^3/N-s for test 1 and 0.08 cm^3/N-s for test 2. Figure 6b shows the open channel test results in comparison to the laboratory jet test results.

Following open channel testing the flow was continued and in situ jet tests were conducted on the channel bed. Water was pumped directly from flow in the channel to the adjustable head tank. Once a test was started, scour readings were taken every 10 minutes using a point gage. Three jet tests were conducted on the channel bed of test 1 and two jet tests were conducted on the channel bed of test 2. Average values of critical stress and the erodibility coefficient were determined from the jet tests run on channel tests 1 and 2: for test 1, τ_c = 2.1 Pa and k_d = 0.12 cm^3/N-s; for test 2, τ_c = 1.1 Pa and k_d = 0.09 cm^3/N-s. The range of the jet test results and the average values are plotted in Fig. 6b, against the results of the laboratory jet tests and open channel tests. The erodibility results from the open channel testing and in situ jet testing are similar and corroborate the laboratory results.

Summary and Conclusions

Jet testing apparatus and methods were described to determine the excess stress parameters k_d and τ_c. An example was given for testing a soil in the laboratory for determining proper placement in the field. A flume test and in situ jet tests were also conducted to verify erodibility. This example indicates the utility of the jet test for determining optimum placement conditions, and for measuring in situ values in the field.

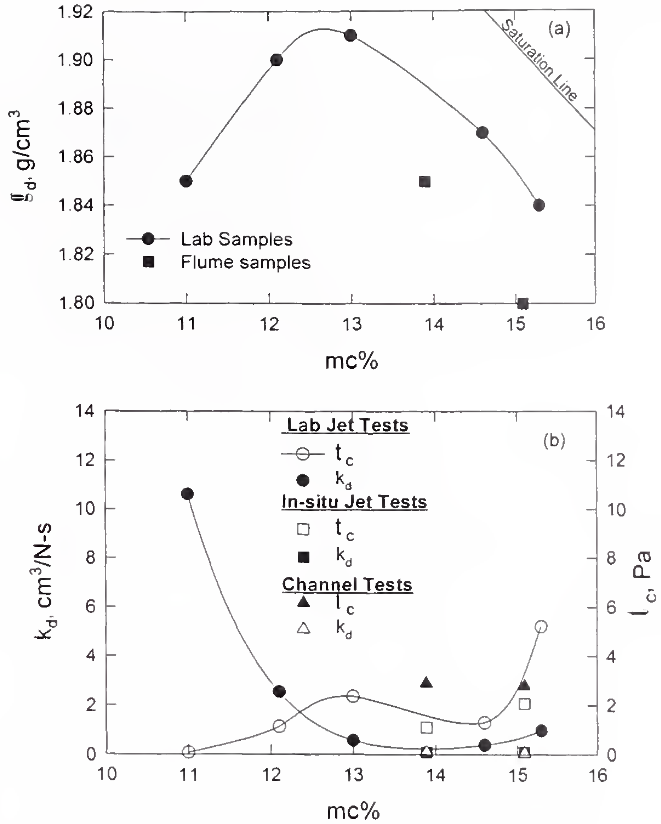


Figure 6. (a) Water content – dry unit weight relationship. (b) Water content – k_d and τ_c relationship for test results.

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Development of Data for a Watershed Model to Predict Agricultural Non-Point Source Pollution

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Usefulness of the BASINS (Better Assessment Science Integrating Point and Nonpoint Sources) Database for Predicting Water Quality and Quantity for a Small Watershed[1]

Abstract

The Hydrologic Simulation Program - FORTRAN (HSPF) is one of the most commonly used hydrologic models. Due to its complexity HSPF requires extensive data input for an accurate simulation. To facilitate use of the HSPF model, the United States Environmental Protection Agency (EPA) has developed a management system which uses the storage capabilities of the ArcView geographic information system (GIS) to store, display, and manipulate a database for the entire United States. This system, called Better Assessment Science Integrating Point and Nonpoint Sources (BASINS), was released in 1996 and includes NPSM (a modified version of HSPF), QUAL2E, and TOXIROUTE. The objectives of our study were to test the usefulness of the BASINS database in predicting runoff from a small tributary watershed and to evaluate a method for improving the precipitation data. A data set for the 103 km^2 Hellbranch Run Watershed in central Ohio was created using BASINS. Precipitation data for the 1991-95 period was used in this study. The first two years of precipitation data were used to establish the initial conditions at the start of 1993. Without any calibration, default BASINS data was used to compare observed and predicted monthly runoff for 1993 which resulted in a Nash-Sutcliffe R^2 of 0.55. To improve this result, a multi-step calibration and sensitivity analysis was conducted. First, ArcView was used to obtain a distance weighted mean precipitation, based on ten weather stations surrounding the watershed, for 49 days where using the weather data in BASINS for a single station nearest to the watershed was perceived to provide the largest influence on the results. The NPSM model was then calibrated by varying eight of the most sensitive process parameters (which are not easy to estimate and are commonly used as calibration parameters) and by statistically comparing the observed and predicted monthly discharge values for 1993. A Nash-Sutcliffe R^2 of 0.87 was achieved in the calibration year and 0.68 was the result for the 1994-95 validation period. A sensitivity analysis of physical parameters (those which might be estimated based on measurements) on monthly runoff was performed using the calibrated model during the validation period. The sensitivity analysis showed that each parameter studied had a small influence on the results. However, the best combination of these parameters resulted in an improved Nash-Sutcliffe R^2 of 0.75 for the validation period. These same calibrated values were evaluated on the calibration period (1993) and the Nash-Sutcliffe R^2 increased from 0.87 to 0.88. Overall, these results are better than most results reported in the literature. This best set of values was then used with the original precipitation data (for a single station that came with BASINS) and the Nash-Sutcliffe R^2 decreased from 0.75 to 0.34. This result suggests that the method used to improve the precipitation data resulted in a substantial improvement in the predicted runoff values. Over the entire three year period, the Nash-Sutcliffe R^2 decreased from 0.82 to 0.66 when the original precipitation was used. We conclude that NPSM should always be calibrated but the BASINS management system provides useful estimates for many of the NPSM input parameters. The next phase of our study is to evaluate the sediment and nutrient simulation. These results will be presented at the 1999 ASAE/CSAE-SCGR Annual International Meeting.

Keywords: BASINS, GIS, HSPF, hydrologic modeling, NPSM

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Introduction

Modeling the environment for better understanding and management of natural resources is an on going process. One of the first comprehensive watershed models was the Stanford Watershed Model created by Crawford and Linsley in 1966 (Hydrocomp Inc., 1998). It has been widely used and has undergone numerous modifications to improve its performance. Crawford and Linsley further developed the model to include sediment transport and water quality simulation which resulted in the Hydrocomp Simulation Program (HSP). Hydrocomp developed the Agricultural Runoff Management Model (ARM) and the Nonpoint Source Pollutant Loading Model (NPS) for the United States Environmental Protection Agency (EPA) in the early 1970's. In 1976, the EPA commissioned Hydrocomp to develop a system of simulation modules in standard FORTRAN that would combine all the functions performed by the HSP, ARM and NPS models. This resulted in the Hydrologic Simulation Program - FORTRAN (HSPF) which due to its complexity requires extensive data input. However, as stated by Hydrocomp (1998): *Although data requirements are extensive and learning to correctly use the model requires some time, EPA recommends its use as the most accurate and appropriate management tool available for the continuous simulation of hydrology and water quality in watersheds.* To assist the model user, the EPA recently developed a watershed management system which integrates a GIS package called ArcView with an extensive database for the entire United States and three models including the Nonpoint Source Model (NPSM), a modified version of HSPF. This system, called Better Assessment Science Integrating Point and Nonpoint Sources (BASINS), was released in 1996 but is undergoing further development due to the high volume of interested users (EPA, 1997).

Because of the scale of the data in the BASINS database, the EPA recommends using site-specific data for studies on small watersheds. This data may not exist, however, in many areas. Gathering the data may be too costly or time consuming and the modeler may have no other choice but to use the data given in BASINS. The purpose of this study was to evaluate the usefulness of the BASINS database for simulating water quantity on a small tributary stream. A technique for improving the precipitation data was also investigated, since precipitation is one of the most important inputs driving the model (Moore et al, 1992).

Model Description and Performance

The HSPF model is a comprehensive, continuous, lumped parameter model that can be used to simulate the movement of water, sediment, pesticides and nutrients on pervious and impervious surfaces, in soil profiles, and in streams and well-mixed impoundments. Spatial variability can be considered by dividing the watershed into hydrologically homogeneous land segments which can be simulated independently with different meteorological input data and watershed parameters (Jacomino and Fields, 1997). The main modeling components of HSPF are the modules that simulate pervious land segments (PERLND), impervious land segments (IMPLND), and route runoff through reservoirs and reaches while simulating in stream processes (RCHRES). Runoff is simulated from the following four components: surface runoff from pervious areas, surface runoff from impervious areas, interflow from pervious areas, and groundwater flow. Algorithms for these processes come from the LANDS subprogram of the Stanford Watershed Model IV (Crawford and Donigian, 1973). Understanding the equations in LANDS is necessary for an accurate calibration of the HSPF model. Both measurable and non-measurable parameters are used in the HSPF model. To avoid confusion, the measurable parameters will be called *physical parameters* while the non-measurable parameters will be called *process parameters*. Due to its complexity, only a brief description of HSPF was given here. However, even with the BASINS interface, extensive research of this model is highly recommended before attempting its use. For a more detailed description refer to the HSPF User's Manual (Bicknell et al, 1996).

The performance of the HSPF model can be considered very good if the difference between observed and simulated values is less than 10 %, good if the difference is 10-15% and fair if the difference is 15-25% (Srinivasan et al, 1998). Laroche et al (1996) found a coefficient of correlation (r) for monthly runoff values equal to 0.9 for the calibration period and a value of 0.93 for the validation period. On a daily time step, Srinivasan et al (1998) found a Nash-Sutcliffe coefficient of 0.55 for a calibration study and 0.57 for a validation study.

Study Area

Hellbranch Run, the site used in this case study, is a 103 km^2 (40 sq. mile) subwatershed of the Big Darby Creek located west of Columbus, Ohio. The Big Darby Watershed is considered one of the most pristine watersheds left in the United States. The immense biodiversity in the area makes it a unique place and, therefore, of great interest to many scientists and environmentalists. Hellbranch Run is located directly to the west of Columbus and is undergoing rapid growth due to the spread of urbanization around the city. However, the landuse in the Hellbranch is still primarily agricultural with only about 10 percent urban land along the eastern boundary where it borders the city of Columbus. The watershed is longitudinal with the streams flowing from north to south. Two small ditches known as Hamilton and Clover Groff join together near the center of the watershed where Hellbranch Run begins. The terrain is fairly flat with only a 35 meter (114 ft) drop in elevation from the top of the watershed to the outlet of the Hellbranch (a distance of about 26.5 km or 87,000 ft). Using various methods, we estimate the time of concentration to be between 7 and 24 hours. The soils in this watershed are predominately of the Crosby series which consists of deep, somewhat poorly drained, slowly permeable soils formed in high-lime glacial till on uplands. Crosby soils are classified in the hydrologic group C. The mean annual precipitation for this area is 99 mm (39 in) with most storms coming from the west. The closest weather station in the BASINS database is located 24 km (15 miles) to the east of the watershed boundary at the Columbus International Airport.

Methods

To simulate runoff, NPSM requires the following data: 1) watershed area, 2) landuse information, 3) reach characteristics (such as lengths, slope, and cross-sectional area), 4) hourly meteorological data, and 5) watershed parameter values. BASINS was used to qualitatively delineate boundaries for the Hellbranch Run subwatersheds (using the elevation and stream network maps as guides) and to determine the percent of pervious and impervious land for each landuse within the specified subwatershed boundaries. The landuse information in BASINS originates from digitized interpretations of aerial photographs that were incorporated with earlier landuse map and field survey information. NPSM was used to add reach information, assign hourly meteorological data, and edit parameter values. The meteorological data from the Columbus Airport station was assigned to all three reaches for the first part of this study. The hourly precipitation was modified using a technique described later in this section. Initial values for most of the parameters were determined for the Hellbranch using rules given by Donigian and Davis (1978) and by a member of the BASINS technical support at the EPA. Using these values gave a worse result then the default values included in BASINS. Therefore, it was evident that the process parameters required calibration and that the defaults given in BASINS were adequate initial values for the calibration process. Initial conditions for the calibration and validation periods were established with two years of simulation prior to each period.

After some primary runs, it was found that the input precipitation was not accurate for several peaks in 1993 which was the year chosen for calibration. The Columbus Airport station was the closest weather station with hourly precipitation. However, daily precipitation from nine surrounding stations (all within a 52 km radius from the center of the watershed) were available from the National Oceanic and Atmospheric Administration (NOAA) and the Ohio Agricultural Research and Development Center (OARDC). These daily values when compared with the daily values included in BASINS revealed large spatial variations during peak storm events. Therefore, a technique for obtaining a more representative precipitation for the Hellbranch watershed was developed. The Spatial Analyst extension to ArcView was used to interpolate daily precipitation from the ten stations over the study area using the inverse distance weighting method. Figure 1 shows the resulting map on the most spatial varying day (7/1/93) where the daily precipitation was 33 mm (1.3 in) less at the midpoint of the watershed than at the airport. Using the confluence of the two ditches flowing into the Hellbranch as the mid-point, a weighted estimate of daily precipitation was determined. Because it was not practical to use this technique for all days in the three year period, only days appearing to contribute the most influence on the results were chosen using two criteria. The first criteria was that the daily precipitation at the Columbus Airport station should be greater than 25.4 mm (1 in) so all days when the precipitation might have been higher at this station then at the watershed would be selected. The second criteria was to select all days associated with an observed runoff greater than 5.7 m^3/s (200 ft^3/s). This resulted in 49 days where the precipitation data was changed, 17 of which were in the calibration period. Since NPSM requires hourly data, the new daily totals were distributed over 24 hours using the original hourly precipitation values as weights.

Once the precipitation dataset was updated, the model was ready for calibration. Studies by Srinivasan et al (1998), Jacomino and Fields (1997), and Laroche et al (1996) have shown that the predicted runoff was most sensitive to these four process parameters: LZSN, INFILT, AGWRC, and UZSN (see Table 1 for descriptions). The

other process parameters in Table 1 have also been found important in hydrologic simulation (Laroche et al, 1996). Each parameter was varied independently to see how it affected the predicted runoff. This information was used to vary the parameters together until an optimum set was found.

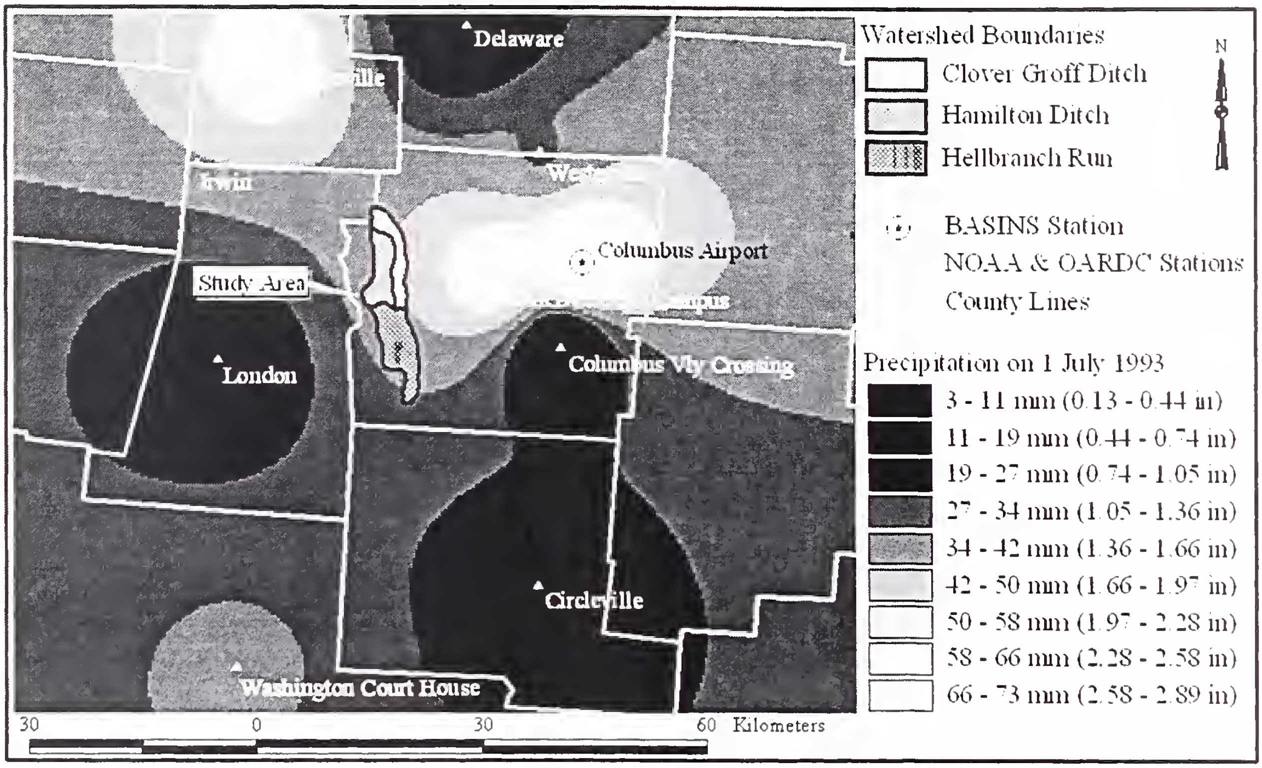

Figure 1: An example of an interpolated precipitation surface generated in ArcView is shown in this map. The precipitation on this day was the most spatial varying out of the 49 days that were changed.

Table 1: Process and physical parameters used in the sensitivity and calibration studies.

Process Parameter	Description	Physical Parameter	Description
LZSN	lower zone nominal storage [mm]	LSUR	length of overland flow plane [m]
INFILT	index to soil infiltration capacity [mm/hr]	SLSUR	slope of overland flow plane
AGWRC	groundwater recession coefficient [day^{-1}]	DEEPFR	fraction of groundwater inflow which will enter deep groundwater and be lost
UZSN	upper zone nominal storage [mm]		
KVARY	parameter which enables the groundwater recession flow to be non-exponential in its decay with time [mm^{-1}]	BASETP	fraction of potential ET which can be satisfied from baseflow
INFEXP	exponent in the infiltration equation	AGWETP	fraction of potential ET which can be satisfied from active groundwater storage if enough is available
INFILD	ratio between the max and mean infiltration capacities		
INTFW	interflow inflow parameter	CEPSC	interception storage capacity [mm]
IRC	interflow recession parameter	PLS NSUR	Manning's n for the overland flow plane of a pervious land segment
LZETP	lower zone ET parameter	ILS NSUR	Manning's n for the overland flow plane of an impervious land segment

After the model was calibrated for monthly runoff using the 1993 data, it was evaluated on the 1994-95 period. Then a sensitivity analysis was performed on the physical parameters (see Table 1 for descriptions) during the validation period. Another calibration was then performed on these parameters still using the 1994-95 dataset resulting in the final best set of values for all parameters studied. This set was run on the full three year period using

both the corrected precipitation as well as the original precipitation so that the results could be compared. The Nash-Sutcliffe coefficient (R^2) and the deviation of runoff volumes (D_V) described by the ASCE Task Committee on Definition of Criteria for Evaluation of Watershed Models (1993) and the coefficient of determination (r^2) were used in our study for monthly values and compared to results found in the literature.

Results and Conclusions

The initial run using the default values resulted in a Nash-Sutcliffe R^2 of 0.55. It was found that by just varying one of the parameters, the correlation could be significantly improved. LZSN had the largest effect by increasing the Nash-Sutcliffe R^2 to 0.76. INFILT and AGWRC also improved the Nash-Sutcliffe R^2 to 0.68 and 0.62, respectively. UZSN did not seem to have any effect on its own, but did significantly improve the Nash-Sutcliffe R^2 when varied with LZSN according to the rules given by Donigian and Davis (1978). Monthly runoff values were not sensitive to INFEXP and IRC at all and, therefore, were left at the default values. A final Nash-Sutcliffe R^2 of 0.87 was achieved in the calibration year by varying all process parameters together (see Table 2 for calibration values). The correlation value dropped to 0.68 during the 1994-95 validation period. Just by varying DEEPFR, however, this value was improved to 0.73. Calibrating all of the physical parameters resulted in a Nash-Sutcliffe R^2 of 0.75 (see Table 2 for calibration values). Using the model with the final best set of values for the calibration period increased the Nash-Sutcliffe R^2 from 0.87 to 0.88. Based on these results, we conclude that NPSM should always be calibrated but the BASINS management system provides useful estimates for many of the NPSM input parameters.

Table 2: The range over which each parameter was studied along with the default value given in BASINS and the optimum value resulting from calibration are listed in this table.

Process Parameter	Range	BASINS Value	Calibrated Value	Physical Parameter	Range	BASINS Value	Calibrated Value
LZSN [mm]	2.5-358	358	127	LSUR [m]	61–97	91	97
INFILT [mm/hr]	0.025-25	4.1	1.0	SLSUR	0.004-0.035	0.035	0.03
AGWRC [day^{-1}]	0.6-0.999	0.98	0.99	DEEPFR	0.0-1.0	0.1	0.18
UZSN [mm]	0.25-127	28.7	17.8	BASETP	0.0-1.0	0.02	1
KVARY [mm^{-1}]	0.0 – 2.0	0.0	0.024	AGWETP	0.0-1.0	0	0.15
INFEXP	0.0-10.0	2.0	insensitive	CEPSC [mm]	0.0-250.0	2.5	0
INFILD	1.0-2.0	2.0	1.0	PLS NSUR	0.001-1.0	0.2	0.212
INTFW	0.0 – 0.75	0.75	0.5				
IRC	$1x10^{-6}$-0.999	0.5	insensitive	ILS NSUR	0.001-1.0	0.1	0.106
LZETP	0.0-0.999	0.1	0.42				

By using the technique developed for improving the precipitation, the total precipitation for the three year period decreased by 11%. The largest absolute and percent reductions, respectively, were from 73 to 40 mm (2.89 to 1.59 in) causing a 45% reduction and from 26 to 8 mm (1.02 to 0.3 in) causing a 71% reduction. The largest absolute and percent increases, respectively, were from 2.5 to 10 mm (0.1 to 0.41 in) causing a 310% increase and from 0.3 to 5.6 mm (0.01 to 0.22 in) causing a 2100% increase. The best set of values was run on the validation period using the original precipitation data that came with BASINS. Figure 2 gives a comparison of the resulting runoff estimated from both the original and area weighted precipitation and the observed runoff for this time period. A reduction from 0.75 to 0.34 for the Nash-Sutcliffe R^2 resulted which suggests that the method used to improve the precipitation data made a large improvement on the predicted runoff. However, when these values were run in 1993, a Nash-Sutcliffe R^2 of 0.94 resulted. This means that in 1993, the original precipitation was a better representation of the actual precipitation then the area weighted values. But over a longer period of time the area weighted precipitation should give better results. This was found by comparing correlations for the entire three year period where the area weighted precipitation improved the Nash-Sutcliffe R^2 from 0.66 to 0.82. Another conclusion from these results is that if the original precipitation had been used to calibrate the parameters, the optimum values would have been very close to the ones found by calibrating on the improved precipitation.

The deviation of runoff volumes (D_V) were calculated for the entire study period for both sets of precipitation data. This resulted in a 2% underestimation using the area weighted values, considered very good according to the

literature, and a 25% overestimation using the original values which is considered fair in the literature. The coefficient of determination was also calculated resulting in an r^2 of 0.86 with the area weighted precipitation and an r^2 of 0.74 with the original precipitation. These values are, respectively, slightly higher and lower than the results from Laroche et al (1996) who found a coefficient of determination for monthly runoff values equal to 0.81 for their calibration study. Therefore, we conclude that the technique used to improve the precipitation data did improve the estimated results.

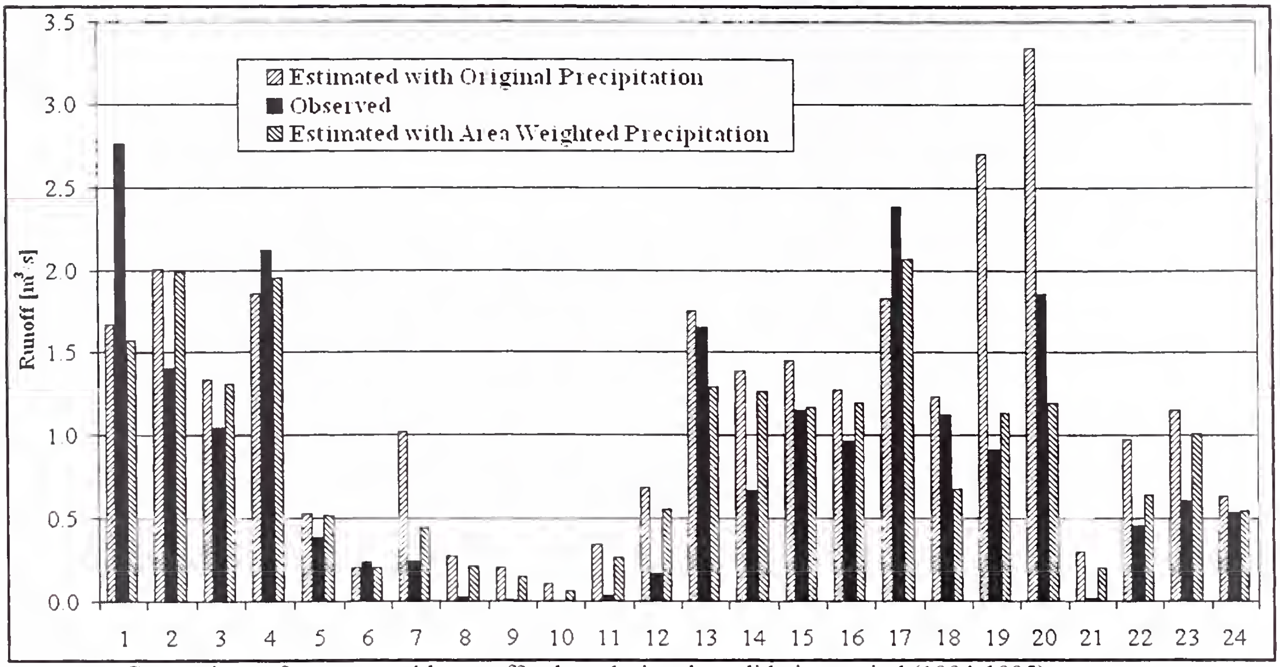

Figure 2: Comparison of mean monthly runoff values during the validation period (1994-1995).

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GIS-Based Lumped Parameter Water Quality Model[1]

Abstract

A GIS based lumped parameter water quality model was developed to estimate the spatial and temporal nutrient loading patterns for lower coastal plain watersheds in eastern North Carolina. The model uses a spatially distributed delivery ratio (DR) parameter to account for nutrient retention or loss along a drainage network. Nutrient delivery ratios are calculated from time of travel and a first-order decay equation for nutrient dynamics. Travel times from any point in the drainage network to the watershed outlet are obtained from simulations using a combined physically-based field hydrology and drainage canal/stream routing model (DRAINMOD-DUFLOW). Nutrient load from contributing areas in the watershed delivered to the main watershed outlet is obtained as the product of field export with the corresponding delivery ratio. The total watershed load at the outlet is the combined loading of the individual fields. Nutrient exports from source areas are either measured, simulated by a mechanistic field scale water quality model such as DRAINMOD-N or estimated from literature data. The lumped water quality model is integrated within a GIS framework with menu interface, display options, and statistical procedures. Within this framework, the model can be used as a screening tool to analyze the effects of different land and water management practices on downstream water quality. Results of the application of the model to evaluate the seasonal and annual export of nitrogen from a large watershed near Plymouth, NC are presented.

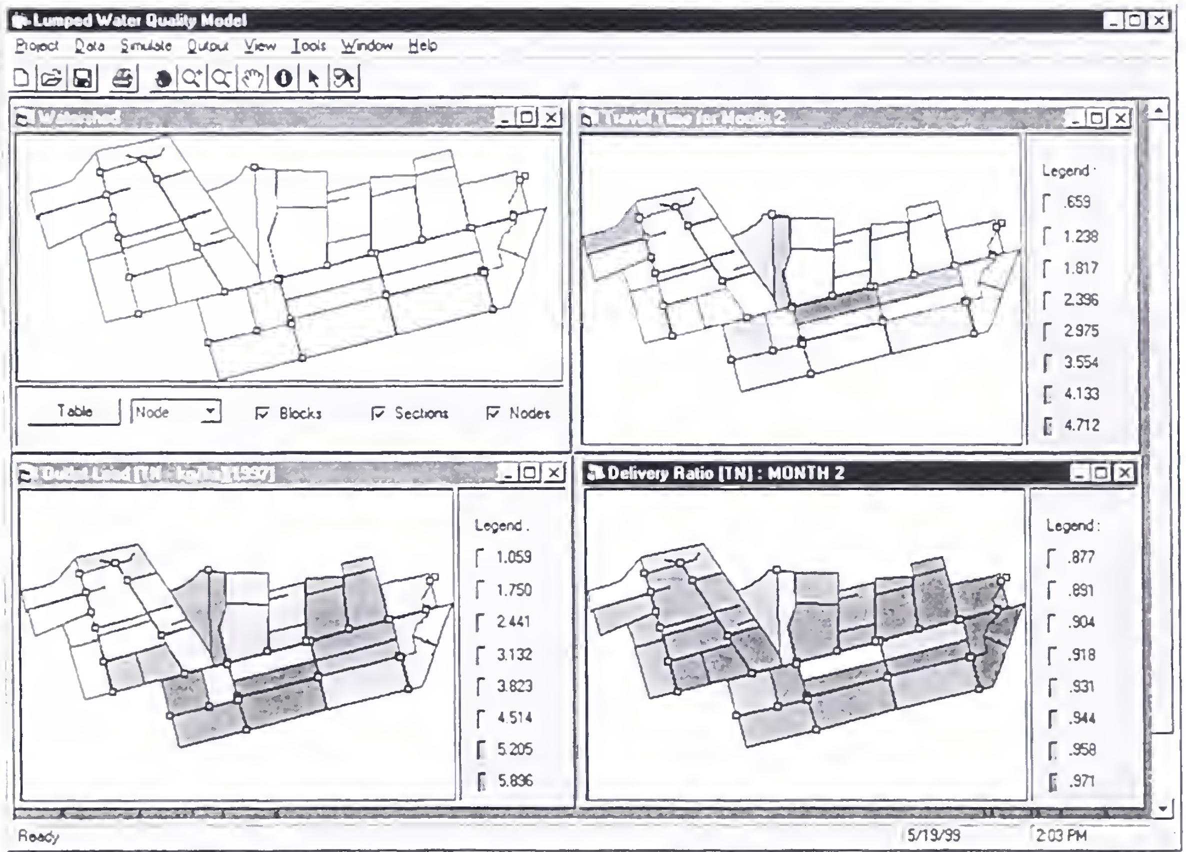

Figure 1. The Lumped Parameter Water Quality Modeling System.

Keywords: Watershed Modeling, Water Quality, DRAINMOD, DUFLOW, GIS

[1] ASAE Paper 992136. Glenn P. Fernandez, Research Associate; George M. Chescheir, Research Assistant Professor; Devendra Amatya, Research Assistant Professor, and R. Wayne Skaggs, Distinguished University and W.N.R. Professor, Department of Biological & Agricultural Engineering, North Carolina State University, Raleigh, NC 27695-7625. Contact author: gfernand@eos.ncsu.edu.

Introduction

Evaluation of the cumulative effects of land use and water management practices on downstream hydrology and water quality of a watershed requires consideration of many factors including climate, geology, soils, vegetation cover, and soil and water management practices. The interaction of these factors is complex and computer simulation models provide a means of integrating contributions from each of them. The spectrum of models used for water quality planning and assessment range from the more comprehensive process-based models (e.g. Ambrose et al., 1981;EPA, 1987) to the conceptual and/or highly simplified lumped parameter models (e.g. Johnes, 1996). Although the complex models are capable of simulating the impacts of the dynamics of natural processes in large watersheds on a shorter time scale, as decision tools for planners and managers they are difficult to use owing to high input data requirements, problems in calibration in large watersheds and parameterization. In addition, the underlying uncertainties in the formulation of processes and parameterization often contribute to uncertainties in predictions (Beck 1987). An overriding consideration in model choice depends upon the user's knowledge of the model, data availability, the system where it is applied and the ultimate application. Most decision makers working with watershed level management may only need planning level information which could be obtained from lumped parameter models with minimal input data requirements and are capable of accurate predictions on longer time scales (Cooper and Bottcher, 1993). Lumped parameter models when coupled with an error and uncertainty analysis can provide decision makers with more information than the traditional deterministic output.

The objective of this study is to develop and apply a lumped parameter water quality model for approximating in-stream processes that affect nutrient and sediment loading at the outlet of lower coastal plain watersheds with poorly drained soils. The lumped parameter water quality model is coupled with a mechanistic hydrology and hydraulics model to predict travel times and, subsequently, delivery ratios for nutrient and sediment loads from source areas within the watershed to the watershed outlet. The model is then integrated within a GIS framework to allow ease of use and to facilitate the pre-and post processing of inputs and modeling results. This study is part of an ongoing large area hydrology and water quality monitoring and modeling study on poorly drained soils in the lower coastal plains of eastern North Carolina (Chescheir et al., 1998).

Modeling Approach

The modeling approach assumes that the net loss in nutrients and sediment loads as drainage water moves from source areas to the watershed outlet is exponentially dependent on time in transit and can be described with a single attenuation coefficient for each constituent. The nutrient and sediment loads delivered to the outlet of a watershed are predicted with empirically determined attenuation coefficient (decay rate). The model is expressed as

$$L = \Sigma DR_i * L_{oi}, \quad i = 1 \ldots n \quad and \quad DR_i = \exp(-kt_i)$$

where DR_i is the delivery ratio for source area i; t_i is the time-of-travel of drainage water from source area i to the outlet; L is the nutrient load at the outlet; L_{oi} is the export coefficient for the source area i or determined as the product of Q_i and C_i , k is an empirically determined attenuation coefficient, and Q_i and C_i are the outflow and drainage water concentration at the source area i, respectively. The delivery ratio is intended to integrate the in-stream transformations and transport characteristics that occurs as the nutrients are transported from the source areas to the watershed outlet. The delivery ratio depends on the time-of-travel along the drainage network and the attenuation coefficient. This model differs from the conventional export coefficient approach (e.g. Johnes, 1996) since it considers the attenuation of the nutrients along the hydrological pathways. Accuracy and uncertainty of the predictions at the watershed outlet depends on the uncertainties of estimating the source loading, attenuation coefficient and the travel time, and of course, on the validity of the hypothesis that the relevant processes can be described by a simple exponential decay function.

Through research conducted on the coastal plain soils over the years (Skaggs and Gilliam, 1981; Gilliam and Skaggs, 1986; Amatya et al., 1998a), the magnitude of nutrient losses (N & P) from agricultural and forested lands and the factors affecting these losses have been determined under various water management (e.g. controlled drainage) and nutrient management practices. Models for predicting nutrient losses at the field edge such as DRAINMOD-N, a nitrogen version of DRAINMOD (Breve et al.,1992), have also been developed and tested on poorly drained soils (Skaggs et al., 1995a, 1995b). Results of these past studies can be used to estimate the nutrient exports needed for the lumped parameter model described above. DRAINMOD based watershed scale models coupled with canal routing component have been developed and recent studies indicate that the integrated models are capable of simulating the hydrology of coastal plain watersheds with poorly drained soils (Amatya et al., 1998b, 1999a; Fernandez et al., 1997).

An important factor that determines the delivery ratio is the time of travel along the stream network. The general procedure for conducting time of travel is to conduct tracer experiments along a stream network by observing the variation in concentrations of the tracer as it moves downstream (Jobson, 1997). Alternatively, the time of travel can be determined by conducting a numerical tracer experiments using a mechanistic hydraulics and water quality model. The linked DUFLOW-DRAINMOD hydrology/hydraulics and water quality model (Fernandez et al., 1997) or DRAINWAT (Amatya et al., 1998b) coupled with a water quality transport model such as WASP4 (Ambrose et al., 1981) can be used to determine travel times.

The model described above is somewhat similar to the one described by Amatya et al., (1999b) using DRAINWAT, but this model is further integrated within a GIS modeling system. The modeling system was developed using Microsoft Visual Basic and MapObjects (Environmental System Research Institute, Inc). The GIS interface was developed to provide an interactive framework for preparing input data sets, editing and creating map layers (e.g. drainage network, soils and land use, computational nodes and reaches, drainage areas), running the simulation and graphically presenting the results through graphs, tables and maps.

Application of Model

Site Description and Methodology

The lumped parameter water quality model described above was tested on 2909 ha drained forested watershed (S4 subwatershed in Fig. 2). The watershed is located in the lower coastal plains in eastern North Carolina. The S4 watershed is part of a larger, heavily instrumented 10000 ha mixed land use watershed (Chescheir et al., 1998). Both organic and mineral soils are present in the watershed. The drainage system of the watershed consists of a network of field ditches generally 0.6 to 1.2m deep and spaced 100 m apart; collector canals 1.8 to 2.5m deep spaced approximately one-half mile which empty into main canals approximately 1.8 to 3.0m deep. Surface cover is characterized by second growth mixed hardwood and pine forest and loblolly pine plantation of various ages and stages (Unpublished data).

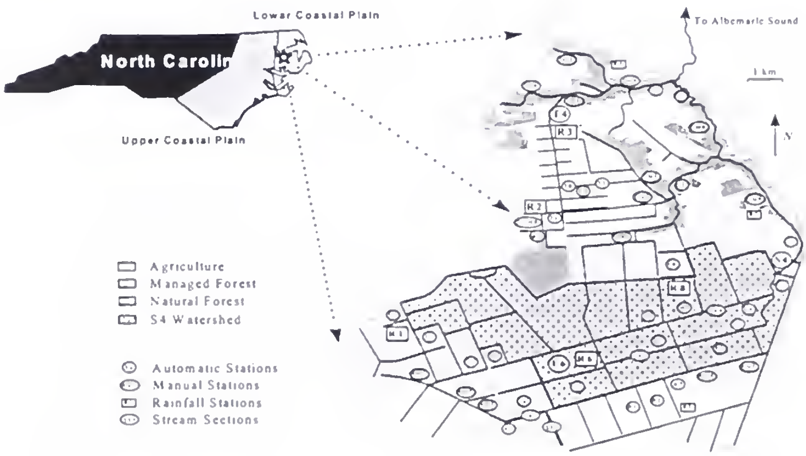

Figure 2. Diagram of study site near Plymouth, N.C.

Flow measurements are being recorded and water quality of the drainage waters are sampled at several gauging and sampling stations within the watershed (Fig 2). The gauging stations are located at five field drainage outlets (F1, F3, F5, F6 and F7), three on the main drainage canals (S1, S2, and S3) and on the outlet of watershed (S4). Instrumentation at the automatic stations includes sharp crested 120° V-notch weirs, water level recorders, automatic samplers and microprocessors to store data and control the samplers. Field and canal sampling stations are serviced biweekly, at which time grab samples are also collected to determine water quality at all stations. More detailed description of the network of monitoring stations for both flow and water quality sampling for this subwatershed and the larger watershed is described in Chescheir et al., (1998).

For modeling purposes, the watershed was divided into 27 fields with the drainage network discretized into 46 segments consisting of 39 canal reaches and 7 weir control structures. The fields are assumed homogeneous with respect to soils, surface cover and water management practices. Field areas, stream lengths, field and canal bed elevations and dimensions of canals and weir control structures, were obtained from field surveys and organized into

GIS coverages and database. Drainage outflows for each field were predicted by DRAINMOD. For hydraulic routing, drainage outflows from each field were treated as inflows into the network at designated nodal points. Flows from the field outlets were routed on an hourly time step to the watershed outlet using the linked DUFLOW-DRAINMOD hydrology and hydraulics model (Fernandez et al., 1997). Predicted flows at the outlet were summarized into mean monthly daily flows for use in the water quality model. To account for the spatial variability of rainfall over the watershed, the rainfall measured at three stations (R3, R6, and R8 gages) within the watershed were distributed to the different fields using a 'nearest-neighbor' approach. The spatial distribution of rainfall over the watershed is intended to account for the observed gradient in annual rainfall from the western edge to the eastern part of the watershed. This trend parallels the orientation of the main drainage canal of the watershed (west to east drainage flow path). In the absence of measured soil water characteristics for all the fields, properties of the dominant soil series for each field were obtained from published values as reported in Skaggs and Nassadzadeh-Tabrizi (1986) and Amatya et al., (1998a). Water quality data collected from biweekly composite and grab samples from five experimental fields in the S4 watershed were used to generate the export coefficients and concentrations for the individual fields. The daily loads calculated from the observed flow weighted concentrations were aggregated to monthly loads and were used as input to the lumped parameter model. The data from the experimental fields were distributed to the 27 fields in the watershed based on similarities in soil type, water management practice, stand age and type of surface cover.

The time-of-travel from any point in the drainage network to the watershed outlet were determined from a numerical tracer experiment. Daily loads from each field were conservatively routed to the watershed outlet using the DUFLOW-DRAINMOD model. Each field was simulated one at a time to determine the travel time from each field. Loading response curves predicted at the outlet during the main flow periods were determined and summarized. Travel times, corresponding to the time when 50% of the input load arrives at the outlet, were obtained from the response curves and were related through nonlinear regression analysis to the routed discharge from the given field, the total upstream drainage area of the field and the length of the flow path to the watershed outlet. The attenuation coefficients used were determined from simulation results using the DUFLOW-DRAINMOD model with a simplified in-stream process model. The in-stream process model includes process descriptions for nitrogen cycling such nitrification, mineralization, denitrification and net nutrient flux exchanges between the water column and the sediment layer. Average delivery ratios obtained from a two-year (1996-97) calibrated run were used to infer a lumped attenuation coefficient for each nutrient fraction. The attenuation coefficients obtained and, subsequently, used in the lumped parameter model were 0.025 (NO_3-N and Organic N) and 0.05 (NH_4-N).

Measured Data

Over a 23-month period (February, 1996 to December, 1997, measured flows at S4 started in Feb 25, 1996), the cumulative drainage at the outlet was 622 mm (Fig. 3). Seventy-six percent of this total flow (476 mm) occurred in 1996 and the remaining 24% (146 mm) in 1997. A large percentage of the flows during the late summer to fall months in 1996 were associated with tropical storms. There was a considerable difference in flow patterns between the two years. In 1996, approximately 75% (359 mm) of the flow occurred during the fall and winter months, 19% (90 mm) and 6% (28 mm) during the spring and summer months, respectively. In contrast, 63% (92 mm) and 35% (50 mm) of the annual flow in 1997 occurred during the winter and spring months, respectively. The trend in nitrogen loads at the outlet of the watershed is similar to the trend in drainage outflows (Fig. 3). In 1996, sixty-one percent of the annual nitrogen load occurred during the fall months with the spring and summer months contributing 28% and 6%, respectively. In 1997, ninety-seven percent of the annual nitrogen load occurred during the winter and spring months. Over the two-year period, 83% of the total watershed nitrogen load occurred in 1996 and the remaining 17% during 1997. Total nitrogen load for the two-year period was 52% nitrate-nitrogen and 6% ammonium-N. Discharge is a strong predictor of the nutrient fluxes with high nutrient fluxes typically occurring during periods corresponding to the dominant hydrological activity within the year. Analysis of the mean daily loads and corresponding drainage from the experimental fields (F1, F3, F5, F6, & F7) showed significant relationships (not shown) between mean daily discharge and mean daily load.

Simulation Results

The temporal trend and magnitude of daily flows predicted by the linked DUFLOW-DRAINMOD model is in good agreement with the observed flows at S4 (Fig 3). Over the 23-month period, the absolute prediction error was 0.05 mm/day, an over-prediction of 5%. Annual drainage outflow in 1996 was over-predicted by 10% (0.14 mm/day) and under-predicted in 1997 by 10% (0.04 mm/day). To a large extent, the over-prediction in 1996 was due to the over-prediction of peak flows during the occurrence of tropical storms (late summer to fall months). The

under-prediction in 1997 can be attributed to potential errors in estimating PET and watershed rainfall during the late winter and spring months. Similar results for a 3-year (1996-1998) period were obtained by Amatya et al., (1999b) for the same watershed using DRAINWAT.

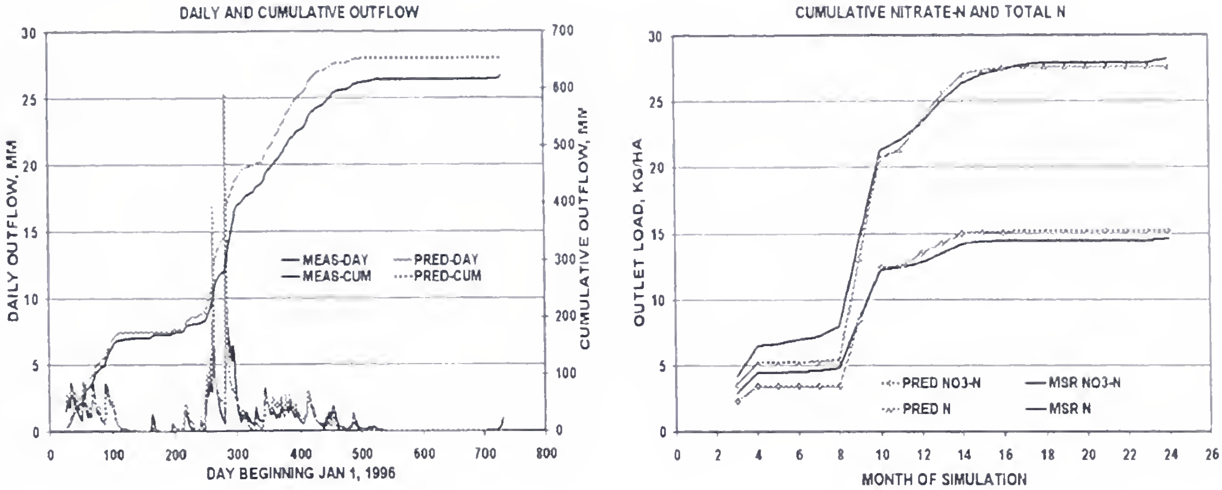

Figure 3. Measured and predicted outflows, nitrate-N and total N for S4 watershed.

Over the two-year period, the trend in the prediction of nitrogen loads at the watershed outlet was similar to the results for outflows. Over-prediction of outflows in 1996 resulted in over-prediction of total nitrogen load by 1% (1% under-prediction for TKN, 6% over-prediction for NO_3-N). In 1997, the under-prediction in outflows resulted in under-prediction of total N load by 19% (25% for TKN and 8% for NO_3). Overall, the prediction errors for total watershed load for two-years were 4% for NO_3-N, -6% for TKN, 1% for NH_4-N and -2% for total N. Ammonium-N was poorly predicted in both years (31% in 1996 and -55% in 1997). The dependence of the load predictions on flow is not surprising. Outlet loads were calculated from export concentrations that were converted to equivalent load by multiplying with the mean flow for the given time period and attenuated with a delivery ratio. The dependence is nonlinear since the delivery ratio is nonlinearly dependent on k and t_i.

Sensitivity analysis indicates that the predicted outlet load is highly sensitive on flow (Q_i) and field export concentrations (C_i) and less on the attenuation coefficient (k) and time of travel (t_i). The normalized sensitivity coefficient calculated for both Q_i and C_i is 1% while for k and t_i is 0.04%. These coefficients indicate that a one-percent change in either Q_i or C_i will result in a corresponding 1% change in output load. On the other hand, a one-percent change in either k and t_i will result in only 0.04% change in output load. Predictions of the total watershed load appear to be less sensitive to the choice of k and t_i. However, the uncertainty of the model output is not directly related solely to the sensitivity of the parameters. A highly sensitive parameter that is known with certainty may have a much less effect on the total uncertainty of the model outputs than a much less sensitive parameter that is highly uncertain (Melching et al., 1996). Of the parameters of the model, the attenuation coefficient k, is probably the most uncertain. This parameter integrates the rates of the processes that describe the cycling of the nutrient within the stream network. Time of travel and flow rates can be reliably predicted with integrated watershed scale models such as the DRAINMOD based models (Amatya 1998b,1999b; Fernandez et al., 1997). Nutrient exports and concentrations from poorly drained soils under various water and nutrient management practices can be reasonably determined from field and/or modeling studies.

Summary and Conclusion

A watershed scale lumped parameter water quality model integrated within a GIS framework (Fig. 2) was developed and tested using two-years of hydrology and water quality data from a 2909 ha forested watershed in eastern North Carolina. The water quality model uses a spatially distributed delivery ratio to approximate in-stream processes as the nutrient is transported from source areas to the watershed outlet. The delivery ratio is dependent on time of travel along the drainage network and on an attenuation coefficient. Results of the application of the model showed that reasonable and accurate prediction of the nutrient loads delivered to the outlet of the watershed requires accurate predictions of the hydrology and hydraulics of the system and characterization of the attenuation of the nutrients in the drainage network. Initial sensitivity analysis indicates that predicted outlet loads are sensitive to the

flow predictions. Although, model predictions are not very sensitive to the attenuation coefficient, the uncertainty in estimating this parameter will affect the overall uncertainty of the model predictions considering that this coefficient lumps all the processes that would otherwise describe the in-stream cycling of the nutrient along the drainage network.

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Assessment Tool for Evaluation of Agricultural NPS Pollution Control Policies in Watersheds[1]

Abstract

An integrated, policy assessment tool is under development within the Unix/NT computer environment. The tool evaluates, at the watershed level, the water quality and economic impacts of alternative agricultural nonpoint source (NPS) pollution control policies. Based on user-supplied policy goals, the tool evaluates specified alternative policy implementation strategies and helps the user assess each one's likely impacts and costs. Both farm and watershed level constraints (e.g., land, labor, capital) and objectives (e.g., profit maximization, risk minimization, reduced environmental impact) are incorporated into the tool. This allows modeling of spatial diversity, both physical and economic, throughout the watershed. The assessment tool consists of a number of integrated modules, linked through a distributed control system. This integration tool lets executable modules built in different languages and running on different machines work together across the Internet. Such structuring enables modifications and additions to be made efficiently to any portion of the tool with minimal effect on the rest of the system. There are six main modules of the assessment tool. The ***Control Module*** acts as a user interface and coordinates execution of the tool's modules, maintaining a shared data store for the other modules during each run of the tool. The ***Data Module*** provides initial data and, in general, provides the interface between the system and data stored in either relational database systems or GIS. Policy goals and policy alternatives for achieving those goals are specified and quantified in the ***Policy Alternatives Module*** based on user descriptions. The ***Farm Planner Module*** selects preferred management alternatives for each farm, based on farm level constraints and farmer objectives. The ***NPS Pollution Module*** estimates water quality constituents at the field, farm, and watershed levels resulting from the preferred management alternatives. Finally, the ***Policy Decision Module*** evaluates attainment of the policy goals for the watershed under the varying preferred policy alternatives. The assessment tool supplies its results to the user as quantitative, decision support information. The decisionmaker can use this information as an aid in determining which policy alternatives best achieve the desired goals. The presented technology will enable watershed-level decisionmakers to compare potential economic and environmental impacts of varying agricultural pollution control policies in specific watersheds. This comparison can aid the decisionmakers in achieving NPS pollution control goals in a cost-effective manner for watersheds.

Keywords: nonpoint source pollution, policy, decision support, simulation, economic analysis

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Problem Statement

Environmental policy for agriculture attempts to encourage farm behavior that furthers a safe and adequate food supply, a quality environment, sound public health, and sustainable agricultural communities. Determining how agricultural policies can best achieve these goals is difficult because policymakers have incomplete information on the impacts of policies. Economic and environmental impacts of policies depend on how farmers respond to policies as well as the spatial distribution of farm economic and physical characteristics. Both are often incompletely known.

Making environmental policy decisions for agriculture should involve the following steps [Fig. 1]: (1) define watershed problems based on a resource inventory; (2) develop clear policy goals (e.g., reduce nitrate pollution from agriculture in the Chesapeake Bay by a stated percentage); (3) generate alternative implementation options to achieve the goals (regulations, tax incentives, cost-sharing options, etc.); (4) assess or predict the water quality and economic effects of implementing each alternative; (5) choose the best alternative through a multicriteria decision process; (6) implement the selected alternative and monitor the effects of implementation on the resources to determine if the desired effects are achieved. The process can then begin again at step (1).

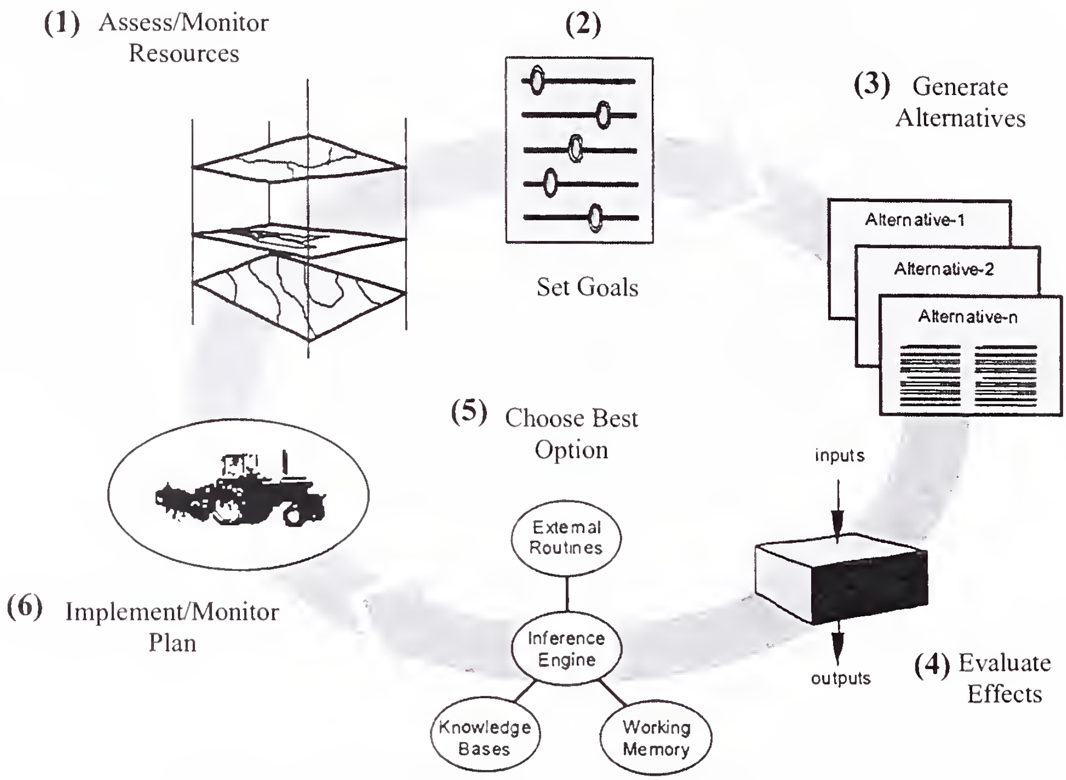

Figure 1. Landuse Planning Cycle; modified from Stone (1995).

In practice, steps (3) and (4) are problematic for at least two reasons: 1) the impact of policy tools is indirect – they affect farmers' behavior which in turn affects resources; and 2) the water quality impacts of changes in farming practices on watershed resources depend on the location and physical characteristics of farms within the watershed. For simplistic cases in which a policy requires a specific agronomic practice to be adopted, it is possible to assume behavior and employ computer models to assess the impacts of the policy on resources and economics. However, the impact of policies is seldom so explicit. More generally, policy implementation provides a set of incentives or penalties designed to encourage a change in behavior by farm managers. How farmers respond to these incentives, disincentives, and constraints determines what practices will be implemented on the farm and, depending on climatic, topographic, and soil variability, what range of impacts may occur on resources. Existing watershed assessment methods do not adequately account for constraints farmers face in responding to water quality policies on their farms, nor the relationship between farm decisions and watershed environmental results.

The goal of this research is to develop a Watershed Assessment Integrated Toolbox (WAIT) for evaluating selected farm policy alternatives so decisionmakers can develop NPS pollution control programs that minimize taxpayer, farmer, and consumer costs while meeting water quality goals. The research objectives are to:

1. Design WAIT as a tool for policy evaluation that supports the Landuse Planning Cycle presented in Figure 1;
2. Implement WAIT using a modular design, enabling additional scientific models and information tools to be incorporated into the assessment tool as needed;
3. Demonstrate the use of WAIT for evaluating alternative nutrient management policies in a specific watershed.

Watershed Assessment Integrated Toolbox

Based on the research objectives, WAIT is designed as the computer portion of a decision support framework that evaluates the water quality and economic impacts of NPS pollution control policies. WAIT aids the decisionmaker in determining the best policy alternative(s) to achieve particular goals. Farm and watershed-level constraints (e.g., land, labor, capital) and objectives (e.g., profit maximization, risk minimization, reduced environmental impact) are incorporated into the analysis. The tool is innovative in that it accounts for spatial variability, both physical and economic, within watersheds in evaluating and selecting between alternative NPS pollution control policies. Thus, it addresses the current research need for spatial consideration of environmental and economic aspects involved with policy implementation (National Research Council, 1993).

WAIT uses interaction and visualization technology to involve users in the decision process. Interactive interfaces for the modules allow users to see and change default values, prompt users for additional information, and display output graphically. These techniques make the users aware of the steps within the decision support process and discourage operation of the model without awareness of its physical representation (i.e., whether or not the steps or the outcome make physical or logical sense for the particular situation).

WAIT is being developed within the Unix/NT computer environment. The flow chart below [Fig. 2] shows the modules that comprise WAIT. Each module is essentially a black box as far as the whole system is concerned. Modules interact indirectly by sharing and manipulating data structures in the Control System's memory. This facilitates modification of individual modules without affecting the whole system. The individual modules are described in the following subsections. The left column of Figure 2 shows the location of the WAIT within the decision support framework. The other columns depict increasing levels of detail within WAIT; the action of the module inside each bolded box follows the cycle indicated by the arrows. For example, the Control Module first calls on the Policy Alternatives Module, followed by the Farm Planner Module and then the Policy Decision Module.

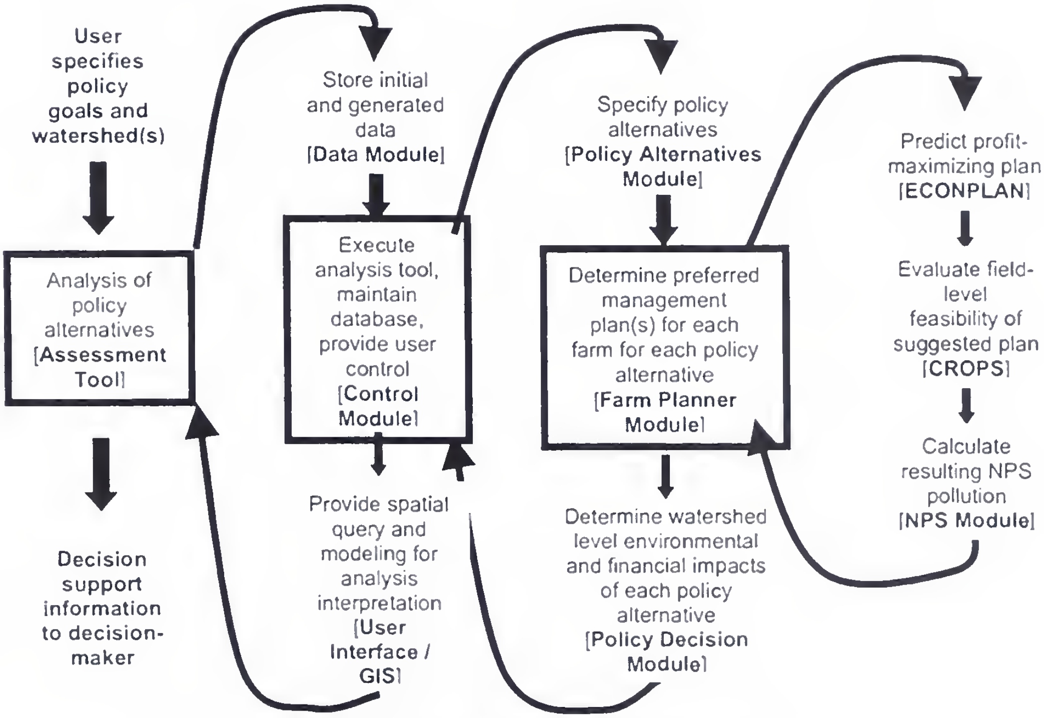

Figure 2. Flow Chart of WAIT Functions.

WAIT addresses the six steps of the decision support framework [Fig. 1], with particular emphasis on steps three and four. We assume that an initial inventory of resources has been done before WAIT is invoked. Within WAIT, the inventory is stored in the Data Module, completing the first step, *Assess/Monitor Resources*. The second step in

the framework, *Set Goals*, is facilitated by the Policy Alternatives Module, and requires input from the user. Goals are established through discussion among the policymakers and the people affected by the policies. This step might involve public meetings to identify problems and desired outcomes. Within WAIT, goals are recorded and a list of quantifiable objectives for each goal is specified. The third step, *Generate Alternatives*, is the primary task of the Policy Alternatives Module. Users specify one or more alternative policy implementation plans or *policy instruments*, defined as sets of performance or practice standards, each with its own set of targeting criteria (defining in which farms, fields, or situations the standard applies), costs, incentives, and constraints. These alternative policy instruments must each be evaluated in the fourth step, *Evaluate Effects*. Evaluation is accomplished by two modules that first determine what practices (BMPs, for example) will be installed as a result of each policy alternative, and then simulate the environmental and economic impact of those practices on the watershed of interest. For an identified alternative, the Farm Planner Module determines the optimal management plan(s) for each farm. To do this it calls on ECONPLAN and CROPS. ECONPLAN predicts the profit maximizing plan at the farm-level. CROPS evaluates the field-level feasibility of the suggested plan and generates a schedule of farming practices for each field of each farm based on the optimal plan. Subsequently, the NPS Module calculates the NPS pollution resulting from each management plan. The fifth step, *Choose Best Option*, is addressed by the Policy Decision Module by generating comparative statistics showing how each alternative policy instrument fared in achieving the user's goals as specified in step two. WAIT supplies its results to the user as quantitative, graphical and spatial decision support information. The decisionmaker can use this information as an aid in determining which policy alternative(s) best achieves the desired goals. The sixth step in the cycle is to implement and monitor the plan. Ongoing evaluation is crucial during the life span of the implemented plan. Changes in the economy or environment, as well as unforeseen interaction can necessitate adjustment. Thus, the cycle should be repeated on a regular basis to ensure the needs of the watershed and its inhabitants are being met in an optimal fashion.

Control Module

The Control Module acts as the user interface of WAIT and coordinates execution of the tool. It is active throughout the use of WAIT because it is the module that knows how and when to call other modules to lead the user through the landuse planning cycle described above. Each of the other modules is registered with the control module so that the latter knows how to execute each module, what data the module require, and what data the module produce. The Control Module also provides a user interface, allowing modules to ask for user input during their execution, if necessary. This module controls the system flow using an expert-system-like goal-directed reasoning system. Its ultimate goal is to provide the user with assessments of the user's alternative policy instruments. Before that can be done, those instruments must be defined and assessed. So, in general, the Control Module will first call the Policy Alternatives Module to establish the goals and policy instruments. This backward-chaining style of control helps to ensure that the system can complete its analysis, even if some part of the analysis fails. That is, if a piece of information that normally comes from a simulation model is missing (i.e., the simulation failed to execute), the data might also be available from the Data Module as a default.

Modules are registered with the Control Module through the Module Integration Tool (MIT). MIT is a component of the Control Module that helps module authors define the inputs, outputs, constraints, and user-interface needs of their modules. Because all data come from the Control Module and are returned to the Control Module, parameter consistency is assured among modules. Other components within the Control Module implement the distributed architecture of the system and the calling of remote modules. There is an execution object created for each module and a communications object that together handle execution of communication among modules through sockets across the Internet.

Data Module

The Data Module provides storage for persistent data including initial data for the other modules. It also implements a generic interface between relational database management systems, geographic information systems, and the Control Module. There are three categories of data utilized by WAIT: (1) static data, (2) scenario data/parameters, and (3) site-specific data. Static, or time-invariant, data include information such as elevation and soil mapping units. The parameters associated with each policy alternative and management practice(s) scenario are included in data tables in this module, accessible to the modules through the Control Module. Since these data do not change between runs of the system, they are stored in a database and loaded into WAIT through the Control Module at startup. Site-specific data include values that the user manipulates interactively.

Policy Alternatives Module

The Policy Alternatives Module enables specification and quantification of policy goals and specification of policy alternatives for achieving particular goals. The module then allows the user to define one or more components of the policy alternative to achieve the desired goals. Policy components are described in terms of desired limits on specific pollutants, the specific policy instrument or combination of instruments to achieve pollutant limits, and the method of application of the instrument. The policy components are then targeted to farms based on site-specific criteria such as farm size (acres), enterprise (e.g., beef cattle, dairy, or poultry), proximity to water, and slope. An example policy component might be targeting riparian buffer strips on all dairies along blue-line streams.

Two types of policy goals that can be evaluated with WAIT include those related the reduction of sediment, nitrogen, and phosphorus and those related to associated costs. For example, policymakers may wish to minimize public and/or private costs of achieving a given level of nitrate reduction. With respect to pollutant reduction, goals may vary in terms of the priority given to reducing each pollutant and the amount of pollution reduction desired. Goals may also vary in terms of the location of pollution reduction. Goals related to costs may differ in terms of the relative weight given to public and private costs. Goals may also vary in terms of the emphasis placed on minimizing overall costs of all farmers in a watershed versus imposing costs fairly across all farmers.

Farm Planner Module

The Farm Planner Module determines management alternatives for each farm based on farm level constraints. The module selects the preferred management alternative(s) for each farm for each policy alternative based on the assumed farmer objectives. The Farm Planner consists of two submodules: 1) a farm-level economic optimization planner (ECONPLAN), which estimates crop rotation acreages and net farm returns on a whole-farm basis, with limited consideration of spatial variability in soil resources, and 2) a cropping scheduler (CROPS) (Stone et al., 1992) which uses the economic planner results as a starting point for developing a field-by-field, multi-year plan for scheduling crops in each field of each farm in a manner consistent with profit goals and field- and farm-level resource constraints. If the field-level constraints considered by CROPS cause its crop scheduling to differ considerably from the first-stage economic plan, then iterations of the economic planner will be necessary to consider the CROPS field-level constraints.

On each farm in the watershed, all field acreages are pooled, and soil resources are categorized by one of three slope classes: flat, moderate, or steep. Flat slopes are assumed to be cropland, moderate could be cropland or pastureland, and steep slopes are assumed to be pastureland. Within each slope category, the percentage of each soil type on the farm is calculated, and the one or two most common soil types are chosen to represent all the soils of the slope category. ECONPLAN predicts the set of activities that minimize the cost to satisfy the objective function. These activities allocate limited acreage resources to crop rotation, tillage, and nutrient application alternatives. Selection of crop rotations is subject to farm-level constraints such as requirements on livestock ration production, manure applications and nutrient losses. Farm-level outputs of ECONPLAN include shadow prices of resources, acreages of crops/pasture and associated tillage and nutrient applications, and net farm revenue above cash costs for current practices and for policy alternatives under review.

NPS Module

NPS Module uses the ANSWERS NPS model to estimate water quality constituents at the field, farm, and watershed levels. ANSWERS is a distributed parameter, NPS evaluation system that has been developed to simulate long-term trends in runoff, sediment, N and P losses from watersheds. Model output is available for the watershed outlet and at specified interior points within a watershed.

Policy Decision Module

The Policy Decision Module evaluates attainment of the policy goals for the watershed under varying policy alternatives and calculates public costs of policy alternatives. Public agency costs for selected BMPs are obtained from the data module. The Farm Planner predicts adoption of new BMPs. Costs are summed for the entire watershed within the Policy Decision Module. The module also displays qualitative information about types of costs that are difficult to quantify but which may be important. For example, litigation costs may be important for some types of regulations but are unlikely to be quantified.

Development of this module requires consideration of the module users, generally policymakers and NPS pollution control program implementers, and how their policy goals can be represented as quantifiable decisions.

Additionally, the module must take into account the formulation of decision rules involving policy goals. Policy goals may be formulated in terms of a single objective over a one-year planning span, as a single objective over a multi-year planning span, and as multiple objectives over a one-year or a multiple-year planning span.

With the single-year, single objective assessment, the performance of each policy alternative is described in terms of its achievement of the most important policy goal over a one-year horizon. For example, the policymaker's objective may be to minimize the sum of private and public costs of achieving a reduction in NPS pollution for the watershed. Costs are estimated over a one-year planning period. The decision module ranks policy alternatives in terms of total public and private costs. The type of solution procedure varies depending on the needs of the user and the type of problem. The user may want simply to compare the level of performance of two alternative policies with respect to policy goals. In this case, the decision module compares performance levels of policy goals from each policy alternative using appropriate output formats such as tables and map displays.

Alternatively the user may wish to select the policy alternative which maximizes achievement of the most important goal when a large number of policy alternatives exist. For example, the program manager may wish to maximize reductions in N runoff in the watershed by installation of new BMPs. The manager must decide how to allocate a limited amount of cost share dollars to encourage adoption of BMPs. In this case, the task of the policy decision module is to allocate the limited funds among competing types and locations of BMPs. Mathematical programming is used to maximize achievement of the objective (N runoff reduction) subject to watershed constraints (limited cost-share dollars).

With the multi-year, single objective assessment, alternatives are ranked in terms of their performance over a multiple year planning horizon. If the most important goal is minimization of public and private costs, alternatives are ranked in terms of the present value of their costs over the assumed planning horizon. Private farm costs are estimated in the farm planner over the same planning horizon.

With the multi-objective assessment, user-specified weights are assigned to competing goals. For example, the user may wish to minimize public and private costs but give more weight to public costs. If the user wishes to compare a few policy alternatives, the module can give a simple display of outputs of each alternative including the weighted achievement of the goals. If a large number of policy alternatives is to be compared, goal programming can be used to find the best policy alternative in terms of achievement of the weighted goals.

Summary

Choosing an agricultural policy that best addresses current environmental and economic farmer concerns is complicated by incomplete information regarding the impact of policy implementation on the area of interest. Farmers' interpretation of and behavioral responses to the policy as well as physical and economic spatial diversity influence the effectiveness of policy implementation. The decisionmaking process for selecting policy alternatives to meet environmental goals can be described by the six-step landuse planning cycle. However, the applications of several steps within this cycle are problematic.

A computer-based tool, WAIT, is being developed as an integral part of an environmental decision support framework. WAIT is comprised of a number of modules that address different areas of the six-step cycle. In particular, computer technology is being used within WAIT to address and resolve problematic steps within the cycle. WAIT accounts for spatial diversity within the watershed of interest. Also, WAIT enables user interaction and, thus, can accommodate policy plan modifications resulting from implementation impacts.

The design of WAIT, as stated in the first research objective, is complete. The modules of WAIT are currently being computer-coded and prepared for integration into the tool. After the modules are combined, further work will be necessary to fine-tune communication among the modules. A demonstration watershed has been identified and baseline data for this watershed are being collected and prepared for use with WAIT.

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Validation of the Subsurface Tile Flow Component in the SWAT Model[1]

Abstract

Nitrate-nitrogen from commercial fertilizers and animal waste applications has been detected in the surface and groundwater sources in many agricultural regions of the U.S. In many Midwestern U.S. watersheds it is critical to simulate subsurface tile flow in order to accurately simulate the hydrologic balance and nitrate transport. SWAT (Soil and Water Assessment Tool) is a large watershed/river basin scale model designed to predict the impact of land and water management, including fertilizer management, on water supply and water quality. In this study, the subsurface tile component of SWAT was validated using plot data from Iowa State University's Northwest Research Center at Nashua, Iowa. Three treatments were selected for validation with varying land use, tillage and fertilizer management. Continuous corn and corn-soybean rotations were grown with combinations of chisel plot and no-till, and a conventional fertilizer treatment. Mean monthly tile flow and nitrate measurements from 1993-1997 were in general within 25% of simulated values. R^2 values from monthly regression analysis ranged from 0.63-0.72 for tile flow and 0.32-0.56 for nitrates.

Keywords: hydrology, watershed model, tile flow, nitrates

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Introduction

Nitrate-nitrogen from commercial fertilizers and poultry and animal manure applications has been detected in the surface and ground water sources in many agricultural regions of the country including Iowa. The current practices of fertilizer application methods and rates are believed to be contributing significantly in the contamination of surface and ground water. Therefore, it is imperative that tillage and planting systems, regarded as best management practices for agricultural sustainability, minimize the potential for chemical runoff and leaching losses to groundwater with alternative management systems. If the potential for contamination is not reduced by developing and successfully demonstrating the innovative nitrogen management practices, the potential for contamination will remain and could result in additional regulations. Because of these concerns, researchers must develop alternative farming practices with the goals of reducing the input costs, and preserving the resource base for the sustainability of agricultural productions systems and protecting the environment.

Many studies have documented nitrate-N concentrations exceeding 10 mg/L in subsurface drain effluent from cropland (Baker, et al., 1975; Drury et al., 1993; Kladivko et al., 1991; Milburn et al., 1990). The mass of nitrate-N lost in drain effluent tends to increase as fertilization rates increase (Angle, et al., 1993; Baker and Johnson, 1981; Bergstrom and Brink, 1986). Losses often exceeded 30 kg/ha/yr when fertilizer was applied in excess of crop needs (Logan, et al., 1980). Bergstrom and Brink (1986) and Gast et al. (1978) measured annual nitrate-N losses of 91 kg/ha and 120 kg/ha from 200 kg N/ha and 448 kg N/ha application rates, respectively. With lower application rates (<100 kg N/ha), Milburn and Richards (1994) measured nitrate-N losses of 10 to 30 kg/ha and annual flow-weighted nitrate-N concentrations of 2 to 5 mg/L.

Although nitrate-N loss in drain effluent tends to increase with nitrogen fertilization rate, Chichester and Smith (1978) found that applied nitrogen accounted for only 25% of nitrate N leached from lysimeters. Mineralized nitrogen accounts for a large percentage of nitrogen leached from soil or used by crops. Less than 50% of fertilizer nitrogen is typically recovered by corn grain (Timmons and Cruse, 1990). Corn may use three to six times more soil nitrogen than fertilizer nitrogen (Reddy and Reddy, 1993).

Model Description

SWAT is a complex, conceptual model with spatially explicit parameterization. Major model components include; weather, hydrology, soil temperature, plant growth, nutrients, pesticides, and land management. A complete description of SWAT model components is found in Arnold et al. (1998). A brief description of the hydrology/water balance is given here. The local (HRU) water balance is represented by four storage volumes: snow, soil profile (0-2 m), shallow aquifer (typically 2-20 m), and deep aquifer (>20 m). The soil profile can be subdivided into multiple layers. Soil water processes include infiltration, evaporation, plant uptake, lateral flow, and percolation to lower layers. Percolation from the bottom of the soil profile recharges the shallow aquifer (ground water recharge). SWAT simulates total ground water recharge as: (1) water that passes past the bottom of the soil profile, (2) channel transmission losses, and (3) seepage from ponds and reservoirs.

Surface runoff from daily rainfall is estimated with a modification of the SCS curve number method. The curve number varies non-linearly from the 1 (dry) condition at wilting point to the 3 (wet) condition at field capacity, and approaches 100 at saturation. A provision for estimating runoff from frozen soil is also included. If rainfall is available at a sub-hourly time step, the Green and Ampt infiltration model (King and Arnold, 1999) can be used to estimate surface runoff. In this study, the daily rainfall/curve number approach was used since sub-hourly rainfall was not available at sufficient spatial detail.

The soil percolation component of SWAT uses a storage routing technique to predict flow through each soil layer in the root zone. Downward flow occurs when field capacity of a soil layer is exceeded if the layer below is not saturated. The downward flow is governed by the saturated conductivity of the soil layer. Upward flow may occur when a lower layer exceeds field capacity. Movement from a lower layer to an adjoining upper layer is regulated by the soil water to field capacity ratios of the two layers. Percolation is also affected by soil temperature. If the temperature in a particular layer if 0° C or below, no percolation is allowed from that layer.

Groundwater flow contribution to total stream flow is simulated by creating a shallow aquifer storage (Arnold et al., 1993). Percolation from the bottom of the root zone is recharged to the shallow aquifer. A recession constant, derived from daily stream flow records, is used to lag flow from the aquifer to the stream. Other components include evaporation, pumping withdrawals, and seepage to the deep aquifer. The model offers three options for estimating potential ET—Hargreaves (Hargreaves and Samani, 1985), Priestley-Taylor (Priestley and Taylor, 1972), and Penman-Monteith (Monteith, 1965). The Penman-Monteith method was used in this study and requires solar radiation, air temperature, windspeed and relative humidity as input. Daily values of wind speed, relative humidity,

and solar radiation were generated from average monthly values. The model computes evaporation from soils and plants separately (Williams et al., 1984). Potential soil water evaporation is estimated as a function of potential ET and leaf area index (area of plant leaves relative to the soil surface area). Actual soil evaporation is estimated by using exponential functions of soil depth and water content. Plant water evaporation is simulated as a linear function of potential ET, leaf area index and root depth and can be limited by soil water content. It is assumed that 30% of total plant uptake comes from the upper 10% of the root zone and roots can compensate for water deficits in certain layers by using more water in layers with adequate supplies.

Lateral Subsurface Flow

Lateral subsurface flow in the soil profile (0-2 m) is calculated simultaneously with percolation. A kinematic storage routing technique that is based on slope, slope length, and saturated conductivity is used. SWAT incorporates a storage-discharge model developed by Sloan et al. (1983) and summarized by Sloan and Moore (1984) which utilizes the kinematic approximation in its derivation. This model is based on the mass continuity equation (mass water balance) with the entire hillslope segment used as the control volume.

The mass continuity equation expressed in mixed finite difference form is

$$\frac{S_2 - S_1}{t_2 - t_1} = iL - \frac{(q_1 + q_2)}{2} \tag{1}$$

where S_1 is the drainable volume of water stored in the saturated zone per unit width ($m^3 \cdot m^{-1}$) at the beginning of the time period. (Width is the dimension not shown in Figure 1.) The drainable volume of water is calculated as the total volume of water minus the amount of water stored in the soil at field capacity. S_2 is the drainable volume of water stored in the saturated zone per unit width ($m^3 \cdot m^{-1}$) at the end of the time period, $t_2 - t_1$ is the amount of time simulated in a time period (h) (for a daily time step, $t_2 - t_1 = 24h$), i is the rate of water input to the saturated zone from the unsaturated zone per unit area ($m^3 \cdot m^{-2} \cdot h^{-1}$), L is the hillslope length (m), q_1 is the discharge from the hillslope per unit width at the beginning of the time period ($m^3 \cdot m^{-1} \cdot h^{-1}$), and q_2 is the discharge from the hillslope per unit width at the end of the time period ($m^3 \cdot m^{-1} \cdot h^{-1}$).

Solving equation 1 for net lateral flow (q_{net}) yields the equation (see Arnold et al., 1998):

$$q_{net} = \left(\frac{2 \cdot w \cdot S \cdot K_s \sin(\alpha)}{\theta_d \cdot L} \right) \tag{2}$$

where w is the width of the hillslope (m), K_s is the saturated hydraulic conductivity ($m \cdot h^{-1}$), and θ_d is the drainable porosity of the soil ($m^3 \cdot m^{-3}$).

When the saturated zone rises so that the water table intersects the upper boundary of the soil layer, water is allowed to flow to the layer above. The amount of flow upward is estimated using the saturated hydraulic conductivity, K_s, and the saturated slope length, L_s.

$$q_{sat} = \frac{K_s \cdot L_s}{L} \tag{3}$$

where q_{sat} is the discharge to the overlying soil layer per unit area ($m \cdot h^{-1}$), L is the hillslope length (m), and L_s is the saturated slope length (m).

Subsurface Tile Flow

When a subsurface tile is present in a soil layer, flow is calculated with the equation

$$q_{tile} = (sw - \theta_d) \cdot (1 - e^{(-24/t_{tile})}) \tag{4}$$

where SW is soil water (mm) and t_{tile} is the tile flow drain time in h. The user inputs depth to tile (d_{tile} in mm) and the model determines the soil layer. There is also a provision for lagging the tile flow to the subbasin outlet with the equation

$$lag_t = 1 - e^{(-1/t_{tlag})} \tag{5}$$

where t_{tlag} is the tileflow lag time in h and lag_t is the tile flow lag coefficient. The amount of flow stored in the subbasin while lagged is estimated with the following equations.

$$q_{tstor} = q_{tstor} + q_{tile} \tag{6}$$

$$q_{tile} = q_{tstor} * lag_t \tag{7}$$

$$q_{tstor} = q_{tstor} - q_{tile} \tag{8}$$

Lagging the tile flow does not effect the total volume, only the timing and thus daily peaks.

Nitrate in tile flow

The amount of nitrate transported by tile flow is estimated by considering the change in concentration.

$$V_{NO3} = q_{tile} - C_{NO3} \tag{9}$$

where V_{NO3} is the amount of NO_3 transported by tile flow and C_{NO3} is the average concentration of NO3 in the layer the flow of q_{tile} for any flow volume using the equation (Williams, 1995):

$$V_{NO_3} = W_{NO_3}(1 - e(\frac{-q_{tile}}{bl \cdot PO}) \tag{10}$$

where WNO_3 is the weight of NO_3 contained in the soil layer, PO is soil porosity, and bl is a fraction of the storage PO occupied by percolating water. The average concentration is

$$C_{NO3} = \frac{VNO_3}{q_{tile}} \tag{11}$$

Field Design and Methods

The experimental site for this study was located at Iowa State University's Northeast Research Center, Nashua, Iowa. This field site was intensively monitored as part of the Leopold Center Project and the ongoing USDA-CSREES MSEA project within Iowa. This study site is located on a predominantly Kenyon silty-clay loam soil with 3 to 4% organic matter. These soils have seasonally high water tables and benefit from subsurface drainage. Pre-Illinoian glacial till units of 200 feet overlie a carbonate aquifer used for water supply.

The study site has 40, one-acre experimental plots with fully documented tillage and cropping records for the past seventeen years. The subsurface drainage system has been in place at this site for more than sixteen years. The tile lines were installed above four feet deep at 95 ft spacing. Each one acre plot has one tile line passing through the middle of the plot another tile line at each of the two borders. The tile lines at the borders help in picking up any cross contamination from the surrounding plots. The tile line installed in the middle of the plot drains about half an acre area. Data on water quality and crop yields were collected at this site from 1993 to 1997. The following treatments of various combination of tillage, crop rotation, and N management systems were established on the 40, one-acre plots.

Treatment 1. Continuous corn with spring chisel plow. Preplant nitrogen application (urea-ammonium nitrate) of 135 kg/ha.

Treatment 2. No till corn-soybean rotation (corn planted in 1993). Preplant nitrogen application of 28 kg/ha with a late spring nitrogen test to determine additional application rate.

Treatment 3. Same as Treatment 2 except a soybean-corn rotation is used (soybeans are planted in 1993).

Each treatment was replicated three times. The N fertilizer (urea-ammonium nitrate UAN) was applied in spring with a spoke injector (Baker, et al., 1989) for single spring and summer side dress applications for late spring N test.

Model Inputs/Calibration

The model requires input on weather, soils, topography and land management. Daily precipitation and maximum/minimum temperatures were input to the model while daily solar radiation, wind speed and humidity were generated from monthly statistics from a nearby station. The Kenyon soil series data was input in the model. The soil was split into four layers with a total depth of 1520 mm. Bulk density ranged from 1.32 to 1.58 and the total available water capacity of the soil is 180 mm. Saturated conductivity averaged 20 mm/hr, clay content 25% and organic carbon 2%. The tile drains were installed at a depth of 900 mm with a design drain time of 24 h (t_{tile}) and a lag coefficient (t_{tlag}).

Table 1. Management Information for Each Treatment

Treatment 1: Continuous corn, chisel plow Constant UAN application			Treatment 2: Corn-Soybean Rotation; No-till; late spring nitrogen test			Treatment 3: Soybean-Corn Rotation; No-till; late spring nitrogen test		
Year	Month/Day	Operation	Year	Month/Day	Operation	Year	Month/Day	Operation
1993	May 14	Elem-N(135 kg/ha)	1993	May 17	Elem-N(28 kg/ha)	1993	May 26	Plant soybeans
	May 16	Chisel plow		May 17	Plant corn		Oct 7	Harvest soybeans
	May 17	Plant corn		July 7	Elem-N(144 kg/ha)	1994	May 2	Elem-N(28 kg/ha)
	July 21	Row cultivator		Oct 21	Harvest corn		May 2	Plant corn
	Oct 25	Harvest corn	1994	May 17	Plant soybeans		June 17	Elem-N(169 kg/ha)
1994	Apr 24	Elem-N(135 kg/ha)		Oct 6	Harvest soybeans		Oct 25	Harvest corn
	May 1	Chisel plow	1995	May 16	Elem-N(28/kg/ha)	1995	May 12	Plant soybeans
	May 2	Plant corn		May 16	Plant corn		Oct 11	Harvest soybeans
	June 2	Row cultivator		June 22	Elem-N(193 kg/ha)	1996	May 21	Elem-N(28 kg/ha)
	Sept 28	Harvest corn		Oct 22	Harvest corn		May 21	Plant corn
1995	May 12	Elem-N(135 kg/ha)	1996	May 30	Plant soybeans		June 24	Elem-N(195 kg/ha)
	May 15	Chisel plow		Oct 8	Harvest soybeans		Oct 21	Harvest corn
	May 16	Plant corn	1997	May 12	Elem-N(28 kg/ha)	1997	May 16	Plant soybeans
	June 14	Row cultivator		May 12	Plant corn		Oct 10	Harvest soybeans
	Sept 22	Harvest corn		June 19	Elem-N(125 kg/ha)			
1996	May 3	Elem-N(135 kg/ha)		Oct 10	Harvest corn			
	May 20	Chisel plow						
	May 21	Plant corn						
	June 24	Row cultivator						
	Oct 21	Harvest corn						
1997	May 12	Elem-N(135 kg/ha)						
	May 12	Chisel plow						
	May 12	Plant corn						
	June 19	Row cultivator						
	Oct 10	Harvest corn						

Inputs to the model are physically based (ie. based on readily observed or measured information). However, there is often considerable uncertainty in model inputs due to spatial variability, measurement error, etc. Two variables were selected for calibration: (1) *esco* - a soil evaporation compensation coefficient and (2) *cn2* - condition II runoff curve number. First, *esco* was allowed to vary between 0.75 and 1.0 with 1.0 signifying no compensation with depth. If simulated and measured flows are more than ten percent calibration is finished. If flow differences continue to exceed ten percent, *cn2* is allowed to vary +(-) 6 to account for uncertainty in the hydrologic condition of the basin. Nitrate in tile flow was calibrated using *nperco*. The *nperco* variable adjusts nitrate concentration in surface runoff relative to the concentration that is allowed to percolate to the second layer.

Validation Results

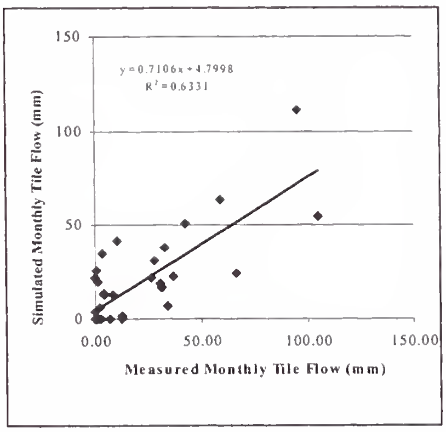
Table 2 shows monthly statistics for tile flow and nitrates for all three treatments. Mean measured and simulated flows are within 8% and R^2 values are all over 0.60. Nitrate monthly means are in general agreement with the model overestimating treatment 3 (no till, corn-soybean rotation).

Table 2. Monthly Tile Flow and Nitrate Statistics

Treatment	Mean Meas Flow	Mean Sim Flow	R^2	Slope	Mean Meas Nitrate	Mean Sim Nitrate	R^2	Slope
Cont Corn; Chisel Plow	16.4	16.5	0.63	0.71	1.90	1.67	0.56	0.80
Corn-Soybean No-till	13.1	13.9	0.61	0.81	1.33	1.66	0.49	0.95
Soybean-Corn No-till	16.0	17.1	0.72	0.82	1.21	2.32	0.32	1.01

Example monthly regression and time series plots for Treatment 1 (chisel plow, continuous corn) are shown in Figures 1 and 2. Figure 1a is the regression plot for monthly tile flow and 1b is the time series graph. Only data from April through November are shown each year since measurements were not taken in winter months. Figures 2a and 2b are regression and time series plots for nitrate in the tile flow (Treatment 1). Although there is considerable scatter in the regression plots, the model is able to match seasonal trends and measured means reasonably well.

Figure 1a-1b. Monthly tile flow regression and time series graphs for continuous corn chisel plow data constant UAN application.



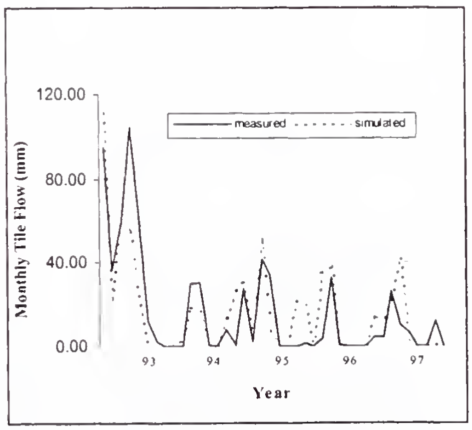
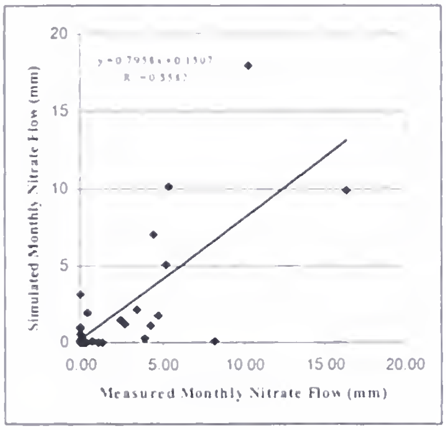
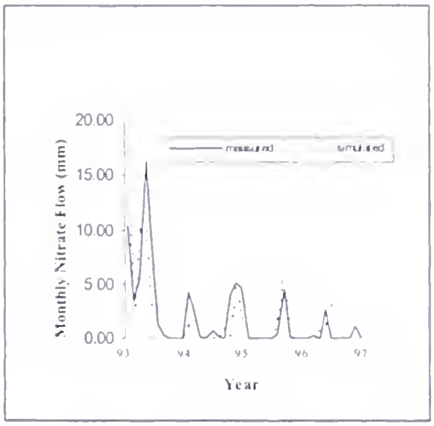


Figure 2a-2b. Monthly nitrate tile flow regression and time series graphs for continuous corn chisel plow data constant UAN application.





Conclusions

With modification, SWAT was able to realistically simulate tile flow and nitrates on plots in Iowa with combinations of continuous corn and corn-soybean rotation, chisel plow and no till, and conventional fertilizer treatments. Validation of the subsurface tile component of SWAT gives greater confidence in model predictions when simulating watersheds and river basins and provides a tool to simulate the impact of tile flow within a basin on regional water supplies, floods, and droughts. Future research will focus on validating the model on a large watershed that is predominately tile drained.

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LAND Model: A Simulation Tool for Estimating the Nutrient Dynamics from Land-applied Animal Manure [1]

Abstract

Historically, nutrient models have not been designed specifically for animal waste land application and do not focus on microbial decomposition processes. The primary objective of this study is to develop a simulation model for predicting nutrient fluxes from the land application of animal waste. The result is the Land Application Nutrient Dynamics (LAND) model, which emphasizes the role of biological processes in the nutrient dynamics of manure decomposition. The simulation model is being developed in STELLA® software focusing on the following processes: 1) microbial decomposition of biomass and organic waste constituents; 2) the plant uptake of nutrients; 3) the volatilization, assimilation, runoff, and leaching of nitrogen compounds, and 4) the adsorption, assimilation, runoff, and leaching of phosphorus compounds. Model structure is based in part on the PHOENIX model for carbon and nitrogen dynamics in grassland soils with modifications for hydrological processes. The phosphorus submodel is partially based on the CENTURY phosphorus submodel modified for a monthly time step. The LAND model is internally linked by a hydrology submodel. Model parameterization and calibration were based on literature values and published data. This model could aid in a better understanding of the physical, chemical, and biological processes that determine the fate and transport of nutrients, mainly carbon, nitrogen, and phosphorus, upon application of animal waste to grassland field systems. This model should also aid in developing effective nutrient management strategies for confined animal feeding operations (CAFOs) and in regulatory policy decision-making.

Keywords: animal manure, nutrients, land application, simulation, model

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Introduction

The concentration of livestock production via large-scale confined animal feeding operations (CAFOs) has led to water quality issues related to nutrient and waste management. Large concentrations of nutrients over a small area can result from animal operations when more animal manure is produced than can be decomposed and assimilated by the available land.

The state of Georgia is presently a primary dairy and poultry producer in the U. S. Last year, the state of Georgia was approached by pork producers to construct large-scale facilities in Taylor and Tattnall counties. Based on the lack of sufficient regulatory policy, controversial environmental problems from CAFOs in other states, and local public opposition, the producers were denied permits by the Georgia Department of Natural Resources (GA DNR). Consequently, Georgia may be losing a potentially valuable economic resource. A simulation tool for assessing and predicting water quality impacts from manure application may assist policy makers, regulators, managers, and farmers in decision-making processes that govern the fate of CAFOs in Georgia, as well as other geographic regions. Currently, simulation models for agricultural practices include the Chemical, Runoff, and Erosion in Agricultural Management Systems (CREAMS) and the Groundwater Loading Effects from Agricultural Management Systems (GLEAMS) models (Leonard et al., 1987), the Agricultural Nonpoint Source (AGNPS) model (Young et al., 1989), and the Riparian Ecosystem Management Model (REMM) (Altier et al., 1993). These models have been validated and deemed useful for various best management practice (BMP) implementation; however, these models are not designed specifically for animal waste land application and do not focus on microbial decomposition processes. The LAND model is being developed to incorporate microbial decomposition processes into the prediction of nutrient fluxes from land-applied animal waste.

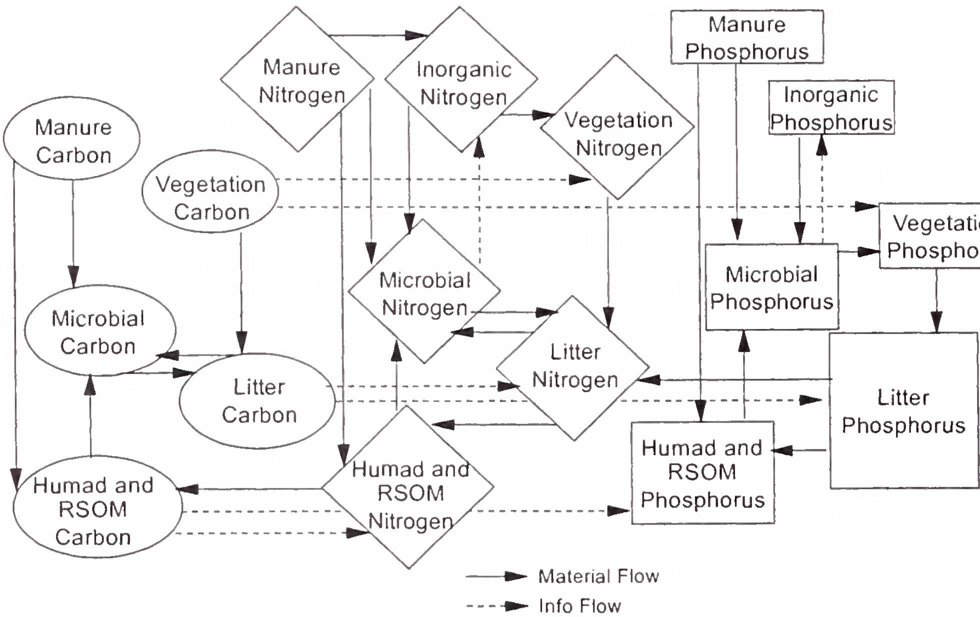

Figure 1. Diagram showing structure of the LAND model.

Model Structure

The simulation model was developed using STELLA® software Version 5.0 (HPS, 1997). The Land Application Nutrient Dynamics (LAND) model consists of carbon, nitrogen, and phosphorus cycling submodels based on the structure of the PHOENIX model for carbon and nitrogen dynamics in grassland soils (McGill et al., 1981) and the CENTURY model for predicting nutrient dynamics in cultivated grassland soils. The PHOENIX model was originally developed to study the interactions between microorganisms and decomposing blue grama grass residues in terms of carbon and nitrogen. The structural and metabolic partitions for biomass described in PHOENIX make it a potentially

useful model structure for the estimation of nutrient dynamics from land-applied animal waste, while the CENTURY model has appropriate corresponding components for the simulation of phosphorus dynamics in soil.

In the LAND model, the major components for each submodel (Figure 1) include a microbial component (bacteria and fungi), a vegetation component (shoots and roots), a litter component, a soil organic matter component [humic and adsorbed soil matter (humads) and resistant soil organic matter (RSOM)], and a manure component. The nitrogen and phosphorus submodels also include respective inorganic components. As in both the PHOENIX and CENTURY models, carbon, nitrogen, and phosphorus are differentiated into structural and metabolic components within the litter and manure components and for the standing dead compartments of the vegetation component. Structural material in manure is typified by cell walls, lignin, and cellulose. Metabolic manure typically consists of cytoplasmic components and membranes (McGill et al., 1981).

Hydrology Submodel

A hydrology submodel links the transport of nutrients between four soil horizons (0-2 cm, 2-6 cm, 6-14 cm, and 14-30 cm deep, respectively). The forcing function of the hydrology submodel is daily precipitation data. Infiltration rate is based on the precipitation data but is limited by the soil porosity (cm^3 void space/cm^3 soil) at the top soil horizon. The vertical transport of soil water from one horizon to the next underlying horizon is driven by Darcy's Law as follows (Jury et al., 1991):

$$Q_{out}{}^{X} = \frac{K_0 A \Delta H}{L} \qquad [1]$$

where,

Q_{out}	=	the soil water flow out of horizon X ($cm^3\ d^{-1}$)
K_0	=	the constant hydraulic conductivity ($cm\ d^{-1}$)
A	=	the cross-sectional area of flow (assumed 1 m^2 or 1 x 10^4 cm^2 in model)
ΔH	=	the change in head (cm)
L	=	the thickness of soil horizon (cm)
X	=	1, 2, or 3, respective to soil horizon.

Also note that due to an assumed water balance, the following is true:

$$Q_{in}{}^{X+1} = Q_{out}{}^{X} \qquad [2]$$

The hydrology submodel produces changes in volumetric soil water content by soil horizon over time, which can be converted to soil water potential given the water retention curve of a certain soil type (Jury et al., 1991). This soil water potential is used to determine the moisture effects of many biological processes within the LAND simulation model. Runoff (R) is calculated on a daily basis as the difference between precipitation (P) and infiltration (INF). Potential evapotranspiration (PET) is calculated using the Thornthwaite equation [Georgia Department of Natural Resources (GA DNR), 1992].

Carbon Submodel

The carbon submodel is composed of five components described previously. For all components, the conserved units for state variables are in $g\ C\ m^{-2}$, the units for flows are $g\ C\ m^{-2}\ d^{-1}$ and for rate constants are d^{-1}, unless otherwise stated. Details on governing equations for the microbial, vegetation, litter, and the soil organic matter compartments are presented in McGill et al. (1981).

The manure component is unique from the model structure developed in the PHOENIX and CENTURY models, but its structure is similar to that of the litter component, being divided into metabolic and structural material. (McGill et al., 1981). The application of manure is the forcing function in this compartment. Manure carbon is immediately partitioned into structural and metabolic compartments by a pre-defined fractional amount, depending on the content in the applied manure. Microbial decomposition results in the transfer of manure carbon to humads and microbial biomass. The rate of metabolic manure decomposition depends on the amount in solution, similar to that of the litter component. Structural manure decomposition also occurs in a similar manner to that in the litter component, including a density effect on microbial decomposition. Note that manure is only applied to the surface horizon, and it is assumed that manure carbon remains in this horizon.

Nitrogen Submodel

The nitrogen submodel consists of components identical to those in the carbon submodel, plus an inorganic component. The rates of transfer between nitrogen submodel compartments and components are proportional to respective carbon submodel rates of transfer by the C:N ratios based on biological nutrient requirements. For all components, the conserved units for state variables are in g N m^{-2}, the units for flows are g N m^{-2} d^{-1}, and for rate constants are d^{-1}, unless otherwise stated. All components except for the shoot compartments of the vegetation component, as well as the manure component, are replicated through all four soil horizons.

The processes of ammonification, nitrification, and denitrification are handled in the inorganic nitrogen component (McGill et al., 1981). A unique feature of the LAND model is that the vertical fluxes of ammonium and nitrate concentrations are driven by the hydrology submodel. The following equation demonstrates the inorganic nitrogen flux from soil horizon X to soil horizon X+1:

$$Q_N^{\ X\ to\ X+1} = Q_{out}^{\ X} \bullet N_C^{\ X} \qquad [3]$$

where,

Q_N	=	the flux of NH_4^+ or NO_3^- from horizon X to horizon X + 1
Q_{out}	=	the soil water flow out of horizon X or the soil water flow into horizon X+1 (Q_{in}^{X+1})
N_C	=	theNH_4^+ or NO_3^- concentration in horizon X
X	=	soil horizons 1, 2, or 3

Note that the inorganic nitrogen flux out of horizon 4 is considered to be leached to groundwater. Runoff inorganic nitrogen concentrations are determined in a similar manner, using the runoff flow calculation described in the hydrology submodel section. The volatilization rate of NH_3 was estimated as a function of temperature and pH (Sherlock and Goh, 1985).

The addition of manure nitrogen to the system is proportional to the rate of application based on the C:N ratio of the manure, and this nitrogen is partitioned into structural and metabolic manure nitrogen in a similar manner as described in the carbon submodel. A pre-defined percentage of ammonium and nitrate allows for the immediate transfer of these inorganic constituents to the respective inorganic compartments.

Phosphorus Submodel

The phosphorus submodel operates much in the same manner as the nitrogen submodel, with most flows being driven based on carbon flows and C:P ratios, with the exception of the inorganic component. This components consists of 4 compartments: 1) labile P, or that which is available for plant uptake; 2) adsorbed P, which is adsorbed to soil; 3) occluded P, and 4) primary P, or that which is naturally occurring in soil. The flows between these inorganic compartments are assumed to be first-order, as presented by Parton et al. (1988).

Model Inputs and Outputs

The "Control Panel" of the LAND model allows the user to define manure characteristics, including: 1) the manure application amount (kg ha^{-1}); 2) the application frequency (applications/year); 3) the fraction of inorganic constituents in the manure; 4) the percentage of total inorganic constituents in total manure mass; 5) the percentage of structural and metabolic organic constituents in the manure, and 6) the soluble fraction of the inorganic nitrogen compounds ammonium and nitrate in the system (not limited to manure). Also, the user may enter precipitation data on a daily basis in cm, plant growth stage for a certain crop, and porosities (cm^3 void space/cm^3 soil) and hydraulic conductivities (k) (cm d^{-1}) for each of the four soil horizons.

Output from all components is represented graphically, including the following categories: Manure, Bacteria, Fungi, Humads and RSOM (resistant soil organic matter), Inorganic Nitrogen, Vegetation, Litter, and Hydrology.

Results

The LAND model was calibrated without the manure component using published results included with the PHOENIX model description (McGill et al., 1981). Next, calibration was conducted with the manure component

Table 1. Parameter Values for Sample Simulation.

Parameter	Value(s)
Hydraulic conductivity (horizons 1-4) (cm d^{-1})	50, 20, 10, 5
Porosity (horizons 1-4) (cm^3/cm^3)	0.6, 0.5, 0.4, 0.4
Application amount (kg ha^{-1})	100
Application frequency (yr^{-1})	20
Soluble Fraction of NH4 in soil	0
Soluble Fraction of NO3 in soil	1.0
Fraction NO3 in manure	1.0
Fraction Inorganic N in manure	0.45
Fraction Inorganic P in manure	0.10

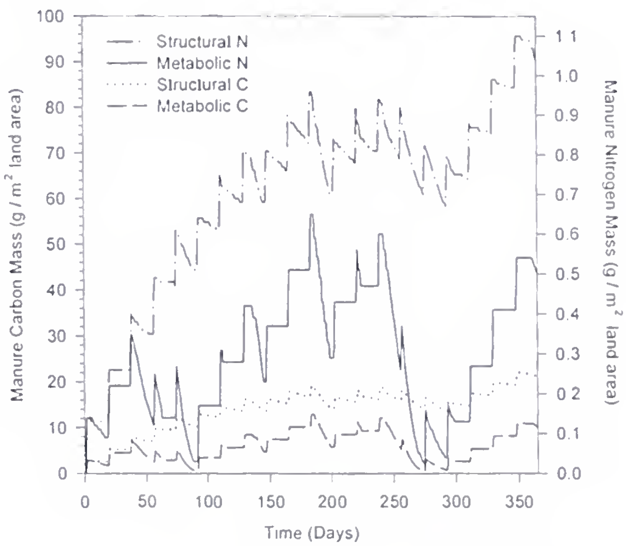

Figure 2. Carbon and nitrogen amounts resulting from simulated manure application.

active, using some parameter values from PHOENIX and some published paramcters for the manure component. Finally, sensitivity analyses were performed by varying parameters and examining the model's response to these changes. Also, manure application rates and frequencies were manipulated to test the model response to these

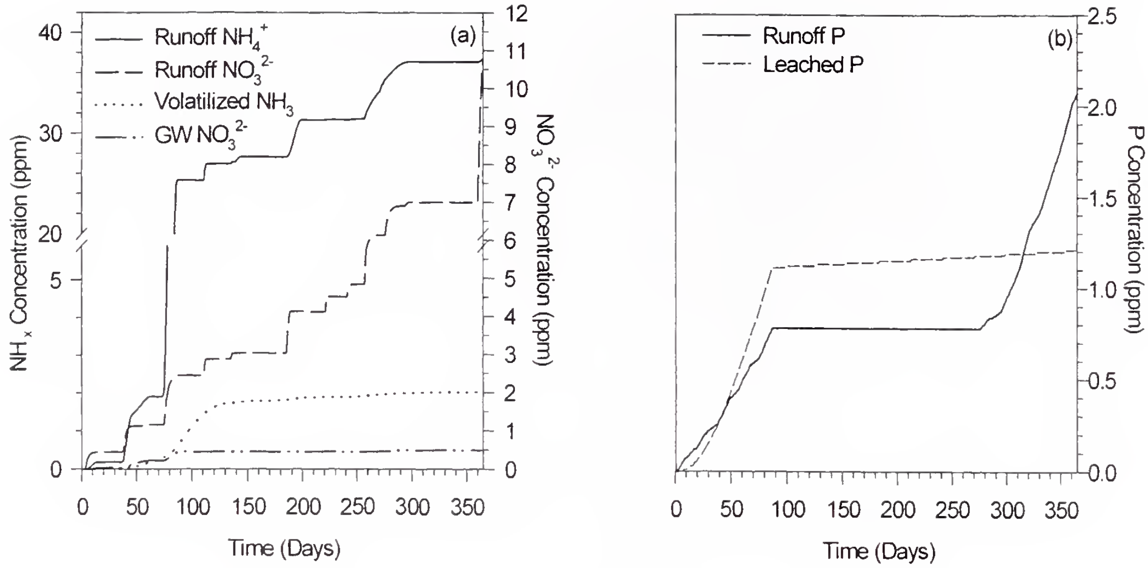

Figure 3. Predicted inorganic nitrogen (a) and phosphorus (b) concentrations resulting from sample simulations using LAND model.

changes, thus evaluating the realistic behavior of the model. Example inputs are given in Table 1, and example outputs are provided in Figures 2 and 3. Calibration and sensitivity analyses have not yet been statistically analyzed.

Discussion and Future Directions

The STELLA® version of a land application model provides a user-friendly interface for manipulating important variables and assessing results. With careful calibration and validation analyses, the LAND model may be developed into a useful research and management tool for animal producers, managers, policy makers, and regulators.

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A Two-dimensional Model for Simulating the Fate of Subsurface-banded Nitrogen[1]

Abstract

Subsurface banding of fertilizer nitrogen (N) offers significant environmental and agronomic advantages over surface broadcast or incorporated N. However, current one-dimensional (1-D) models cannot adequately simulate the fate of banded-N where N movement is two-directional. Hence, a physically-based, field scale, lumped parameter model designed to simulate the fate of banded-N for corn-fallow rotation is presented. The model is comprised of a 1-D moisture sub-model and 2-D N sub-model. The model domain is the root depth × crop row space, discretized into equally-spaced nodes in the vertical and lateral directions. The moisture sub-model applies the implicit finite difference method to Richard's equation (in the vertical direction) and provides inputs to the N sub-model. Actual transpiration, which is applied as an abstraction, is calculated as a function of environmental demand, root depth, and soil moisture availability. The N sub-model applies the alternating-direction-implicit method to the convective-dispersion equation to solve for urea, ammonium (NH_4^+), and nitrate (NO_3^-) concentrations. Urea dissolution is simulated using an empirical relationship. Urea hydrolysis is simulated using the Michaelis-Menten and first-order reactions, with the first order reaction applied at higher urea concentrations, as is expected in band-applied conditions. Urea hydrolysis is stopped at urea concentrations in excess of 400 mg/mL. Nitrification is simulated using both zero- and first-order reactions, as a function of NH_4^+ concentration. At NH_4^+ concentrations above 3 mg/g-soil, nitrification is stopped. While both urea hydrolysis and nitrification are simulated as temperature-dependent reactions, nitrification also accounts for soil pH and moisture content. The moisture sub-model outputs percolation loss from the root zone while the N sub-model outputs NO_3^--N leached from the root zone and N removed by the crop. Model testing and sensitivity analyses focussed on N-related parameters are underway. Potential avenues for model improvement include enabling the model to simulate discretely applied fertilizer pellets and incorporating other crop and crop rotation scenarios.

Keywords: nitrogen, subsurface bands, 2-D model, nitrate leaching

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Introduction

Compared to surface broadcast, subsurface application of nitrogen (N) fertilizer results in lower N losses in surface runoff and through volatilization. Crop N uptake of subsurface-applied N can be further improved by banding as compared to incorporation. In conservation tillage, for subsurface-N application there is no alternative to banding. Given its environmental and agronomic advantages over surface application, subsurface N banding is likely to gain greater acceptance in both conservation and conventional tillage systems.

While undergoing transformations to plant-available forms, subsurface-banded fertilizer-N moves laterally and vertically with respect to the band, a two-dimensional (2-D) situation. Consequently, crop N uptake and N leaching losses from subsurface-banded N are likely to differ from surface application or incorporation where it can be reasonably assumed that N movement is one-dimensional (1-D), in the vertical direction. Currently-available management models (e.g., GLEAMS and OPUS) as well as research models such as SOILN, LEACHM, and ANIMO are not intended to simulate banded-N applications given their 1-D nature. The quasi 2-D DRAINMOD-N model simulates N transport in 2-D only in the saturated zone; hence, its utility in simulating fate of band-applied N in the mostly unsaturated vadose zone is limited. Kaluarachchi and Parker (1988) proposed a 2-D finite element model for N (ammonium and nitrate) transformation and transport under unsaturated conditions. Compared to the analytic solution, the model overpredicted solute concentrations and showed excessive apparent dispersion with a single solute specie, surface-applied in a 2-D domain (Kaluarachchi and Parker, 1988). We have no information on the application of the model by Kaluarachchi and Parker (1988) for simulating subsurface band-applied N.

A 2-D model could facilitate better N management under banded conditions, as compared to a 1-D model. The objective of this paper is to present a computationally efficient 2-D mathematical model for simulating N transformation and transport from band-applied urea ($CO(NH_2)_2$). Model testing and sensitivity analyses on selected model parameters are currently underway.

Model Development

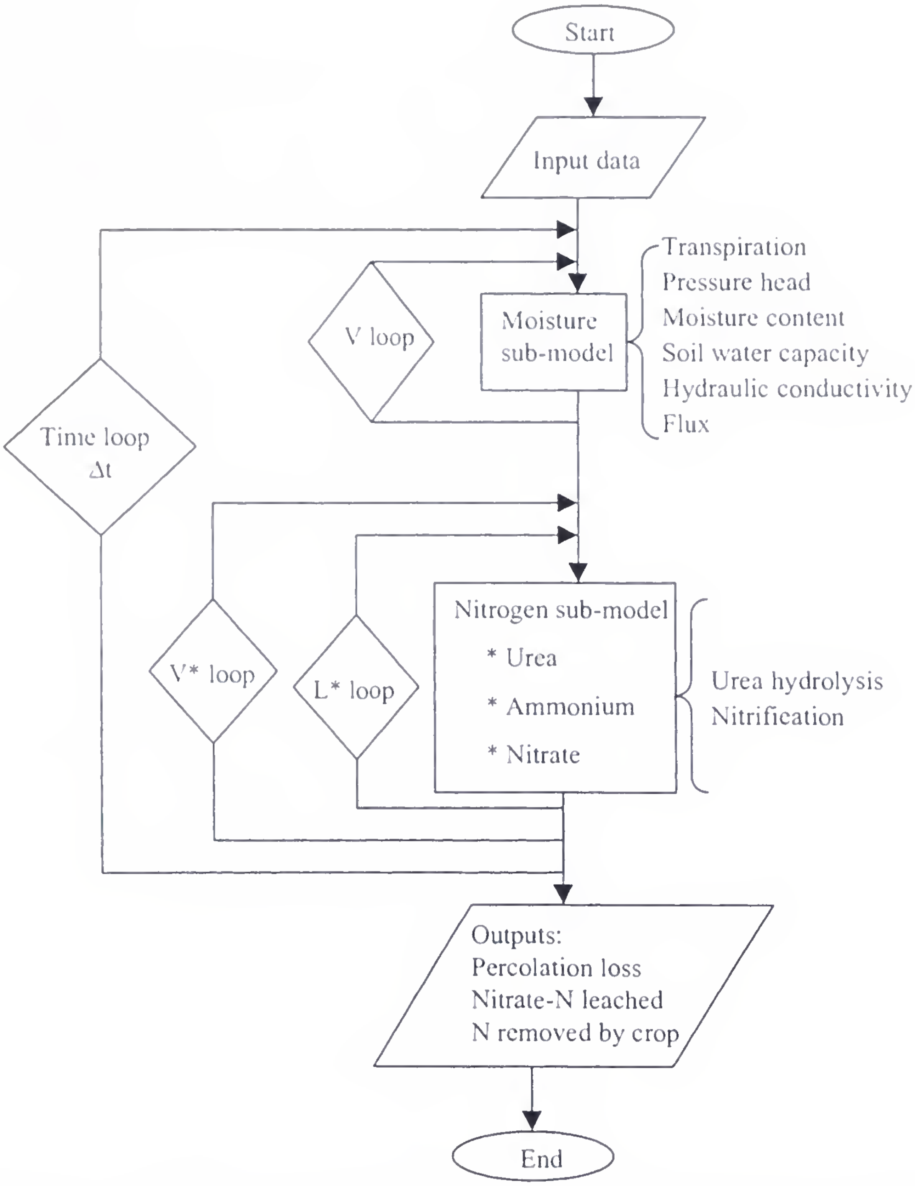
Overview

The model is a physically-based, field scale, lumped parameter model with the root zone × crop row spacing as its domain for a corn-fallow rotation. The model has two components (fig. 1): (1) moisture redistribution sub-model (1-D) and (2) N sub-model (2-D). For moisture redistribution, the root zone is divided into equally-spaced nodes (fig. 2). For N simulation, each vertical layer is further divided into horizontal, equally-spaced nodes. The model uses a variable time step that is boundary flux-dependent. Infiltration is input to the model while transpiration is an abstraction. A unique pressure head (h)-volumetric moisture content (θ) relationship is assumed in the model.

Two urea applications, as starter and sidedress can be simulated. Urea granule dissolution is simulated with an empirical relationship. Urea hydrolysis simulation is concentration-dependent. Nitrification is simulated as a one-step reaction, with the reaction order being concentration-dependent. The inhibitory effects of high substrate concentrations are considered for both urea hydrolysis and nitrification. Ammonia volatilization and N-loss in surface runoff are disregarded since N is subsurface-applied. Nitrogen immobilization due to organic matter and NH_4^+ fixation by clay are considered to be negligible given the reduced soil-substrate contact in band applications. Percolation losses, nitrate-N (NO_3^--N) leaching from the root zone, and crop removal of N are simulated.

Model Inputs and Outputs

Model inputs include soil properties (porosity (f), cm^3/cm^3; bulk density (ρ_b), g/cm^3; longitudinal dispersivity (λ_l), cm; soil pH); soil-moisture parameters by node (soil moisture characteristic curve; saturated hydraulic conductivity (K_S), cm/h; van Genuchten parameters); initial θ by node; weather data (air and soil temperatures (T), °C; relative humidity (R_H), decimal; solar radiation, cal/cm^2-h); residual-N status for all nodes (urea, ammonium (NH_4^+), and NO_3^- concentrations in mg/cm^3); N transformation-related rates (urea hydrolysis and nitrification) and parameters (e.g., urea granule dissolution rate constant); and crop information (planting and harvesting dates; urea application dates; row spacing; maximum root depth; urea application rates; urea application depths; urea application distances from a crop row). Model outputs include percolation losses, m^3/ha; NO_3^--N leached from the root zone, kg/ha; and N removed by the crop, kg/ha.



*L and V in the figure represent lateral and vertical nodes, respectively. In the first half of the time step, for each V node, all L nodes are solved while for the second half of the time step, the order of solution is reversed.

Figure 1. Simplified flowchart of the model with the model outputs shown in bold caption.

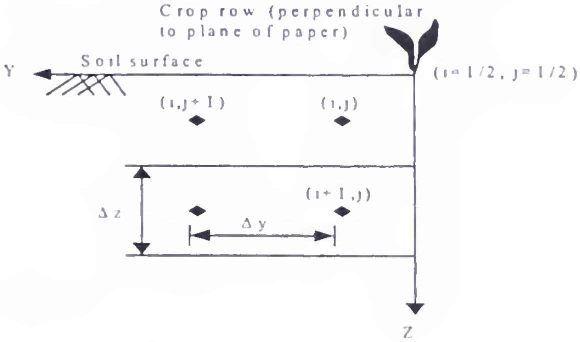


Figure 2. Discretization of the 2-D domain

Finite Difference Formulation and Time Step Determination

Moisture redistribution is simulated by applying the finite difference (FD) scheme to Richard's equation. The implicit method is used to discretize the modeling domain (fig. 2) with L equally-spaced (Δz) nodes in the vertical direction, Z. The implicit method is utilized given its stability and ease of solution (FD equations are easier to linearize and solve) as compared to the explicit (simpler but less stable) and Crank-Nicolson (more accurate but more difficult to solve) methods (Remson et al., 1971). The Douglas-Jones predictor-corrector method is used to linearize FD equations (Remson et al., 1971). The resulting FD equations produce a tridiagonal matrix that can be conveniently solved using the Thomas algorithm (Remson et al., 1971).

The convective-dispersion equation (CDE) is used for the N sub-model (Selim, 1994). The horizontal y axis is divided into M equally-spaced (Δy) nodes; Δy and Δz are kept equal. The alternating-direction-implicit (ADI) method is used to solve the CDE using the Thomas algorithm. For the first half of the time step, a set of M linear FD equations is solved L times while in the second half, a set of L linear equations is solved M times, using a two-step process. The Thomas algorithm can be used in the ADI method because one axis is treated explicitly in the first-half of the time step while the other axis is treated explicitly in the second-half of the time step.

The model uses a variable time step that is calculated as (Feddes et al., 1978):

$$\Delta t^{k+1} < \frac{\varsigma \cdot \Delta z}{\left| q_b \right|^k} \qquad [1]$$

where Δt is time step, h; k is the time step increment; ς is the time step factor (0.02-0.034), with ς increasing with flux; and q_b is boundary flux, cm/h. The time step varies in a range of 0.025-2.000 h, increasing with decreasing boundary flux conditions.

The simulation involves the solution of Richard's equation for $\Delta t/2$ to calculate intermediate $h_i^{k+1/2}$ values using known h_i^k values. The $h_i^{k+1/2}$ values are used to calculate coefficients of the Richard's equation for use in the corrector. The corrector is then used to calculate h_i^{k+1} using h_i^k and coefficients at the k+1/2 time step. Moisture flux and θ values (from the moisture predictor) as well as initial N species concentration ($c_{i,j}^k$) values are used as inputs in the set of FD equations obtained from the CDE for the first-half of the time step to calculate $c_{i,j}^{k+1/2}$. In the second-half of the time step, $c_{i,j}^{k+1}$ values for all three N species (urea, NH_4^+, NO_3^-) are calculated at the end of the time step using the inputs from the moisture corrector and $c_{i,j}^{k+1/2}$ values.

Process Simulation

Moisture sub-model

Soil moisture distribution is simulated using the following h-based form of Richard's equation, suitable for both saturated and unsaturated conditions (Skaggs and Khaleel, 1982):

$$C(h)\frac{\partial h}{\partial t} - K(h)\frac{\partial^2 h}{\partial z^2} - \frac{\partial K}{\partial z}\left(\frac{\partial h}{\partial z} - 1\right) = -T(z,h) \qquad [2]$$

where C is soil water capacity ($\partial\theta/\partial h$), mL/cm^3-h; t is time, h; and T is the transpiration rate, mL/cm^3-h. The FD equations for Eq. [2] are presented in Shah (1999). Moisture flux through the internode (except the soil surface) is calculated using Darcy's equation where the internodal $K_{i+1/2}$ and h_i and h_{i+1} values are used. Internodal $K_{i+1/2}$ is the arithmetic mean of the nodal K_i and K_{i+1} if K_i is within 10% of K_{i+1}, otherwise, the geometric mean is used. The Mualem-van Genuchten method is used to calculate K (de Jong, 1993).

Mixed boundary conditions, involving both Neumann (q) and Dirichlet (θ or h) conditions are applied to the soil surface and bottom of the root zone. Under precipitation conditions, infiltration provides flux at the soil surface. Under non-infiltration conditions, $h_{1/2}^k$, (minimum pressure head under air-dry conditions) at the soil surface at any time k, is expressed as (Feddes et al., 1978):

$$h_{1/2}^k = \frac{RT}{Mg}\ln(R_H) \qquad [3]$$

where R is universal gas constant, 8.314×10^7 erg/mole-K; T is air temperature, K; M is molecular weight of water, 18 g/mole; and g is gravitational constant, 980.6 cm/s^2. Internodal K at the soil surface is calculated for a location $\Delta z/4$ below the surface using the geometric means of K_1 and $K_{1/2}$. Given high head variations close to the soil surface due to crusting, the more conservative K obtained with geometric means is used to calculate evaporative flux at the soil surface with Darcy's equation (Feddes et al., 1978).

At the bottom of the root zone, it is assumed that θ does not vary with depth. However, due to rounding, C can become infinity rendering the solution unstable; hence it is assumed that h_{L+1}-h_L is equal to 0.025 cm. Darcian flux through i = L+1/2 is used to calculate percolation losses from the root zone.

Potential crop evapotranspiration (PCET) is calculated using crop coefficient and reference ET. Potential evaporation (PE) is estimated using the Priestley-Taylor method. Potential transpiration (PT) is calculated by subtracting PE from PCET. Potential transpiration is assigned to each root node using a modified form of the 40:30:20:10 rule; the first, second, third, and last quarters of the root zone extract 40, 30, 20, and 10% of the total water, respectively. Actual transpiration is calculated by taking into account the PT and h value at the node using the method of Feddes et al. (1978). No transpiration occurs from the surface node.

Nitrogen sub-model

The N sub-model simulates N transformations involving urea and NH_4^+ while crop N uptake is simulated for NH_4^+ and NO_3^-. Nitrogen leaching losses are simulated for NO_3^-. Nitrogen movement in the soil profile is simulated by the CDE (Selim, 1994), separately for the three N species:

$$\frac{\partial[(\theta)(c)]}{\partial t} + \rho_b \frac{\partial(S)}{\partial t} = -\left(\frac{\partial(J_y)}{\partial y} + \frac{\partial(J_z)}{\partial z}\right) \pm Q \quad [4]$$

and,

$$J_y = -\theta(D_m + D_t)\frac{\partial(c)}{\partial y} + q(c) \quad [5]$$

$$J_z = -\theta(D_m + D_l)\frac{\partial(c)}{\partial z} + q(c) \quad [6]$$

where c is solute (N specie) concentration in soil solution, mg/mL; S is solute sorbed on soil, mg/g-soil; J_y and J_z are solute fluxes in y and z-directions, mg/cm^2-h, respectively; Q includes both source (+) and sink (-) terms, mg/cm^3-h; D_m is molecular diffusion coefficient of solute, cm^2/h; D_t and D_l are dispersion coefficients for the soil in cm^2/h, respectively; and q is moisture flux, cm/h. The dispersion coefficients are defined as $D_t = \lambda_t v$ and $D_l = \lambda_l v$, where λ_t and λ_l are dispersivities in the transverse and longitudinal directions, in cm, respectively, and v, the pore velocity = q/θ, in cm/h. For zero lateral moisture flux, D_t and q(c) Eq. [5], are both equal to zero. The molecular diffusion coefficient D_m can be expressed as:

$$D_m = D_0 \cdot b \cdot \theta \quad [7]$$

where,

$$b = \theta^{2\ 3} \cdot f \quad [8]$$

where D_0 is the solute diffusivity in bulk aqueous solution, cm^2/h and b is tortuosity, cm/cm. Equation [8] is the empirical Millington and Quirk tortuosity model.

Boundary conditions for the surface layer (i = 1, j = 1,..., M) assume that ${}_uc = {}_ac = {}_nc = 0$ where the subscripts u, a, and n apply to urea, NH_4^+, and NO_3^-, respectively. For the bottom layer (i = L, j = 1, ..., M), ${}_uc = {}_ac = 0$, while $\partial({}_nc)/\partial z = 0$. For the extreme lateral nodes (i = 2, ..., M, j = 1 or L), assume that ${}_uc = {}_ac = 0$, while $\partial({}_nc)/\partial y = 0$.

For urea, Eq. [4] is applied without the sorbed solute term since urea does not adsorb on soil. The urea source term accounts for urea addition due to dissolution of urea granules and is applied to only those nodes were urea is applied. Urea addition due to dissolution is simulated at θ values above wilting using an empirical relationship (Shah, 1999). The urea sink term accounts for urea loss due to conversion to NH_4^+ through urea hydrolysis. For ${}_uc \le 25$ mg/mL, the Michaelis-Menten reaction is applied to simulate urea hydrolysis rate (Bremner and Mulvaney, 1978). For $25 < {}_uc \le 400$ mg/mL, urea hydrolysis is simulated using a first order reaction; for ${}_uc > 400$ mg/mL, urea hydrolysis is stopped (Wetselaar, 1985), a likely occurrence with high urea application rates in bands. Temperature impact on urea hydrolysis is modeled using the Arrhenius equation.

Equation [4] applies to NH_4^+ without modification. The sorbed and dissolved phases of NH_4^+ are partitioned using the Freundlich isotherm:

$${}_aS = {}_aK \cdot {}_ac^{1/n} \quad [9]$$

where ${}_aK$ is the Freundlich slope parameter (mg-NH_4^+/g-soil) and 1/n is the Freundlich non-linear coefficient. Substitution of Eq. [9] in Eq. [4] results in non-linear FD equations. Hence, to linearize the FD equations, while the 1/n term is retained for the known ${}_ac$, it is assumed to be one for the unknown ${}_ac$. The 2-D N model of Kaluarachchi and Parker (1988) showed very little sensitivity to the Freundlich non-linear coefficient.

The NH_4^+ source term is equal to the urea sink term in mg-NH_4^+/cm^3-h. Nitrification is simulated with a first-order reaction rate at ${}_ac < 0.054$ mg/mL while a zero-order reaction rate is used for higher concentrations, similar to

the OMNI model. At soil NH_4^+ concentrations in excess of 3.0 mg/g-soil, nitrification is stopped (Wetselaar, 1985), which is likely in urea bands due to high NH_4^+ concentrations. The nitrification rate is adjusted to account for soil pH, temperature (Arrhenius equation), and soil moisture. The NH_4^+ sink term accounts for NH_4^+ removed by the crop. Ammonium-N removed by the crop is estimated by multiplying the amount of moisture removed through transpiration from a horizontal layer (i, j = 2,..., M-1) with the average $_ac$ in that layer in mg-NH_4^+-N.

Equation [4] is applied to NO_3^- without the sorbed solute term since NO_3^- is held on the soil surface in negligible amounts. The NO_3^- source term is equal to the NH_4^+ sink term, expressed in mg-NO_3^-/cm^3-h. The NO_3^- sink term accounts for NO_3^- removal by the crop and is estimated the same way as NH_4^+ removal. Nitrate-N leaching from the bottom layer (i = L, j = 1, ..., M) is obtained by multiplying the moisture flux at the bottom internode (i = L+1/2) with the average NO_3^--N concentration in the bottom layer (i = L, j = 1, ..., M). The amount of N removed as NH_4^+ and NO_3^- is summed up over the crop season to estimate N removed by the crop.

Model Application

The model will be tested using data available for a research site in Kentland Farm, 16 km from Virginia Tech, Blacksburg, Va. for the May 1998 - March 1999 period. The research site had been planted to no-till corn with control treatment (no N) as well as experimental treatments comprising of urea N applied in subsurface bands. Hourly weather data as well as moisture content and inorganic-N content in the root zone (120 cm) are available for the site. Residual NO_3^- and NH_4^+ from N application will be estimated for the fertilizer treatment by adjusting for background NO_3^- and NH_4^+ levels in the control treatment.

We will conduct sensitivity analyses on this model concentrating on mainly N-related parameters, such as, granule dissolution rate and reaction rates of both urea hydrolysis and nitrification. Model sensitivity to unsaturated hydraulic conductivity parameters will also be evaluated.

Model Improvement

Much of the future work will depend on the results of the sensitivity analysis. However, some of the potential avenues for model improvement are as follow:

1. modify the model to allow for 3-D simulation of discretely-applied fertilizer pellets;
2. develop a physically-based method to simulate the dissolution rate of urea particles;
3. determine reaction orders and rates at very high substrate concentrations, and develop relationships to predict N transformation rates for different soils using key soil properties; and
4. incorporate other crops and crop rotation scenarios.

Finally, after adequate validation, we intend to incorporate this model into an established management model. This will permit better estimation of the fate of applied-N while allowing the modeling of the entire N cycle as well as other nutrients and chemicals. Such a model will be able to serve a larger user base.

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Initial Application of SWAT to Whole Turfgrass Systems[1]

ABSTRACT

Turfgrass systems are one of the most intensively managed land uses in the U.S. The number of turfgrass systems in the U.S. currently exceeds 16000 and is projected to increase by 300 to 400 per year to meet consumer demand. Development of turfgrass systems usually results in a more intensively managed land use. The impact of that management on water quality and quantity is of vital importance. SWAT is a comprehensive watershed scale model developed to predict management impacts on water, sediment, and chemical yields for ungaged watersheds. SWAT allows for subbasin division based on management, soils, etc. Approximately one year of data from a municipal course in Austin, TX was used to evaluate SWAT on whole turfgrass systems. Surface water volumes using the SWAT CN approach did not correlate well with measured values. Improvements and current modifications within SWAT could potentially improve the simulation capabilities for turfgrass environments.

Keywords: turfgrass, modeling, SWAT, hydrology

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INTRODUCTION

Environmentally sound management of golf course turfgrass provides both public and private facilities with environmental, cultural, and economic benefits. According to the National Golf Foundation (1998), there are over 16,000 golf courses operating in the United States with approximately 1.2 new course opening every day. Public demand is increasing for golf course managers to maintain high quality turfgrass on golf courses, but also to protect water and soil resources in the vicinity of these facilities (Balogh et al. 1992; Beard and Greene 1994). Management of existing golf courses and construction of new facilities often is a lightning rod of environmental and water quality concern (Balogh et al. 1992). Water quality, resource allocation, and environmental issues specifically related to turfgrass management on existing and proposed facilities include:

- Excessive use of potable water for turfgrass irrigation in arid and semiarid climates.
- Potential offsite movement of nutrients and pesticides into surface water and groundwater.
- Direct exposure of beneficial soil organisms, wildlife and aquatic systems to pesticides and fertilizers.
- Loss of soil and sediment during golf course renovation, construction, and from poorly drained areas, shaded spots, and high-traffic areas on golf courses.
- Disturbance of the water balance during golf course development or renovation by converting vegetation, changing topography, and increasing the use of water for irrigation.
- Disturbance and loss of wetlands.
- Disturbance and change of existing land use patterns.

Agronomic and water quality problems associated with management of golf courses are readily identified (Balogh et al. 1992). The challenge faced by the golf course superintendents and owners is to select environmentally sound and economically feasible plans to resolve these conditions. Using a risk reduction approach many golf course managers are beginning to use site specific Turfgrass Management System (TMS) plans to maintain high quality turfgrass and protect water and soil resources. TMS plans are integrated best management practices involving irrigation, fertilization, pest and disease control, soil and water conservation practices, and other agronomic practices related to turfgrass management.

Using computer simulation models is a cost effective means of evaluating the relative efficacy of turfgrass management practices (Balogh et al. 1992; King and Balogh 1997, 1999). Computer models combine site specific soils, topographic, vegetation, and climate data with turfgrass management practices to estimate (1) water budgets; (2) local soil erosion; (3) nutrient budgets; (4) fate and movement of pesticides; the water quality impacts of different turfgrass management practices. Prior to implementing specific water conservation or water quality protection plans, computer simulation can be used to assess the relative effectiveness of these practices (King and Balogh 1999). Assessing management practices with computer simulation techniques gives managers the advantage of identifying turfgrass management practices with potentially favorable or adverse effects on the environment.

With varying degrees of sophistication, computer models then predict runoff, leaching, sediment transport, and offsite movement of nutrients and pesticides. The accuracy of predictions from simulation models is contingent on the availability of reliable soils, climate and irrigation data (Balogh et al. 1992). Cost effective evaluation of many of the integrated best management practices (BMPs) for the TMS plans has been demonstrated using field scale computer simulation models (Balogh 1999). Horsley and Moser (1990) used the Pesticide Root Zone Model (PRZM) to evaluate the relative risk of using commonly applied turfgrass pesticides on a golf course on Cape Cod, Massachusetts. Using the PRZM model, Primi et al. (1994) evaluated the leaching potential and maximum concentrations of pesticides in leachate for turfgrass on several soils found on Long Island, New York. Using the results of the simulation pesticides were classified as leachers, transitional compounds, and nonleachers. Smith and Tillotson (1993) used the GLEAMS model to evaluate the leaching potential of turfgrass herbicides (e.g. 2,4-D, dicamba, MCPP, and dithiopyr from a greenhouse lysimeter system. The lysimeter leaching data was compared to simulation results. GLEAMS tended to overestimate the amount of water percolating through the lysimeters and the concentration of herbicide in the leachate. However despite the overprediction of losses, the difference between the actual and simulated data was relatively small.

The Environmental Policy Integrated Climate (EPIC) model developed by (Williams 1995) has been modified specifically for use with turfgrass systems by King and Balogh, 1997, 1999. King and Balogh (1999) demonstrated the effectiveness of using a combination of effluent water for irrigation and modified fertilization practices to conserve water resources and reduce the amount of nitrogen lost in runoff and subsurface leaching from turfgrass. These studies were conducted using data from a golf course in Austin, Texas. In an earlier study, Rosenthal and Hipp (1993) used EPIC to evaluate nitrate and pesticide losses from turfgrass plots on clay soils in Texas. Both field monitoring and simulation demonstrated that nitrate losses and pesticide losses in runoff were highest for high and medium maintenance treatments. Loss of nitrate and herbicides in runoff were significantly reduced in the low maintenance management treatment.

Field scale models are capable of simulating growth and management conditions for individual components on golf courses however, watershed scale models are required to demonstrate the overall efficacy of BMPs for protecting surface water and groundwater in the vicinity of a golf course (Balogh et al. 1992; Cohen et al. 1993). Field scale models project water quality conditions at the edge of the field, e.g. fairway, green, or tee. During a permit application for a water quality evaluation, a whole watershed projection is necessary. Watershed scale models incorporate the spatially variable effects of (1) soils; (2) selective application of water quality BMPs depending on proximity to environmentally sensitive areas; (3) differential water conservation practices; (4) use of vegetative buffer zones used to protect surface water and groundwater, and (5) unmanaged or natural areas occurring on golf courses.

The objective of this work is to evaluate the water balance capabilities of SWAT on a whole turfgrass watershed. The Soil and Water Assessment Tool (SWAT) simulation model has been tailored specifically for turfgrass conditions. SWAT has the advantage of using sophisticated modules for simulation of plant growth, the water budget, nutrient processes, and pesticide processes (Arnold et al. 1996). Unlike other watershed scale models, specifically used for highly variable urban landscapes, SWAT accounts for both the hydrology and turfgrass agronomic conditions. The results of the SWAT simulation are compared to runoff monitoring data from a golf course in Austin, Texas. This study was designed to model a turfgrass system under existing management conditions. The simulation model and the simulation strategy demonstrate a significant improvement in methods to evaluate the efficacy of water conservation and water quality protection practices. These strategies can be used on both existing and proposed golf course watersheds.

METHODS AND MATERIALS

Data Collection

Surface water quantity was collected at Morris Williams Municipal Golf Course, located in Austin, TX from March 20, 1998 to April 30, 1999. Morris Williams Golf Course is an 18-hole golf course measuring approximately 76 ha (190 acres). Morris Williams is located adjacent to the south side of Robert Mueller Municipal Airport between Manor Road and Martin Luther King, Jr. Boulevard.

Surface water from precipitation or irrigation on the course will either infiltrate the surface soils or follow the surface topography. The majority of this runoff will drain internally to one of three primary drainage ways (Fig. 1). The drainage ways transect the course from north to south and merge farther downstream and offsite as Tannehill Branch (U.S. Geological Survey 1988). Surface and subsurface drainage on the course present potential problems. Several areas are subject to periodic flooding and standing water. The course is exposed to run on water from bordering streets and the municipal airport.

A subsection of the site, which, transects the center portion of the golf course was used in this study. A series of grassed waterways, culverts, and casual water detention areas cross this section of the course. The drainage way conducts surface runoff from the course and airport across the middle of holes #10, #15, #16, #17, #2, and #7 (Fig. 1). The topography (Fig. 1) is such that the contributing area of this subsection will include 8 fairways and 10 greens.

Soils on Morris Williams have been subjected to considerable disturbance and redistribution as a result of municipal airport construction projects. The Travis soil (fine, mixed, thermic ultic paleustalfs) occupies the largest area on the site and is the dominant soil on the study site selected.

Mean annual precipitation in Austin, TX is approximately 810 mm (31.9 inches). Mean daily temperatures range from a minimum 3.9 C (39 F) in January to a maximum 35 C (95 F) in July (Texas Almanac, 1997). Austin, TX has 270 growing season days. The golf course is irrigated with potable water from the city and water pumped from a branch of Tannehill Creek. Irrigation is applied on a needed basis determined by course personnel.

The fairways and greens are bermudagrass and are managed at a moderately intense level. Nitrogen is applied as a combination of organic, biostimulants, slow release and fast release N formulations. Mean annual application of nitrogen on all the municipal courses in Austin is 330 $kg \cdot ha^{-1}$ on fairways and 1174 $kg \cdot ha^{-1}$ on greens.

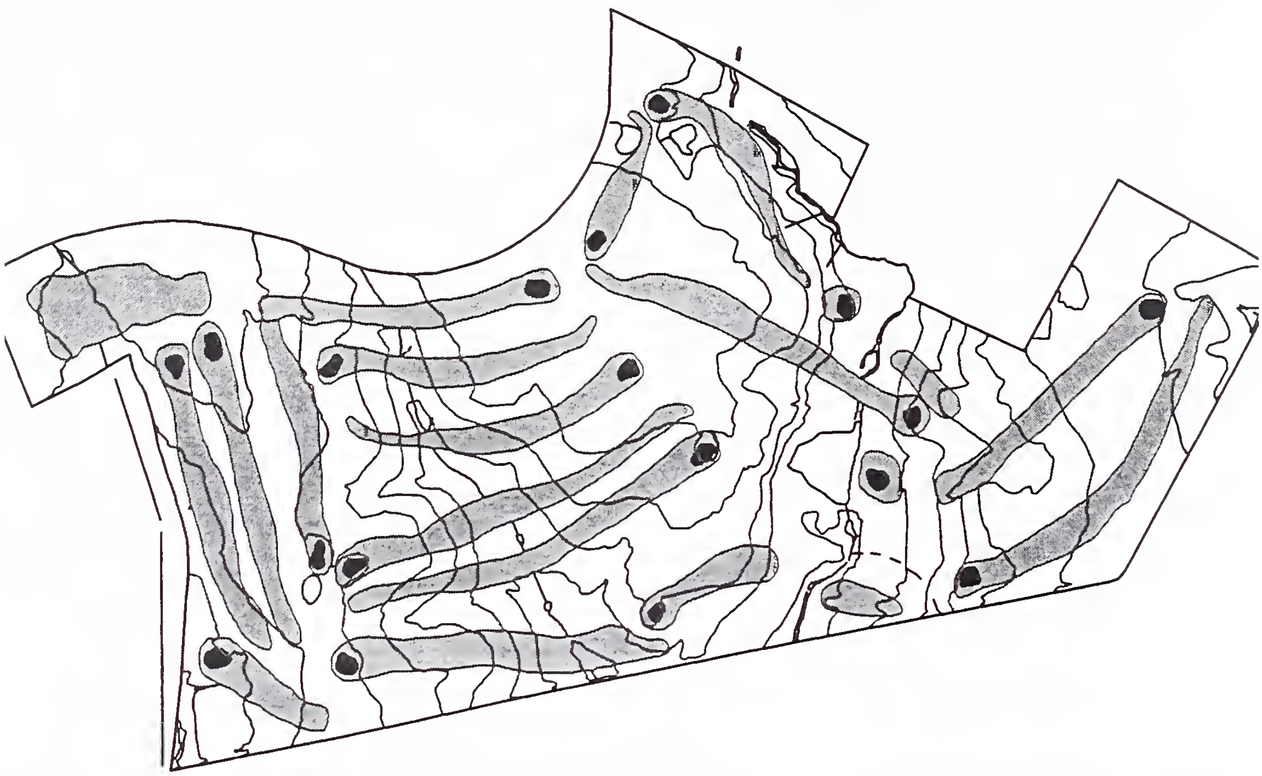

Figure 1. Fairway and green layout of Morris Williams Golf Course with overlay of topographic contours and streams.

Model Selection

The Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998) model is the proposed simulation tool for adaptation to this study. SWAT was selected because it is public domain, user friendly, robust, widely accepted, and validated in an array of geographical locations. Other advantages of SWAT include a comprehensive crop growth model and the ability to vary management schemes and practices throughout the watershed.

SWAT is a comprehensive watershed scale model developed to predict management impacts on water, sediment, and chemical yields for ungaged basins. SWAT currently operates on a daily time step and can simulate long periods of time (100+ years). SWAT validation and application efforts have been completed for water yield (Arnold et al., 1993; Srinivasan and Arnold, 1994; Rosenthal et al., 1994), sediment yield (Bingner et al., 1996), nutrient loss (Jacobson et al., 1996), and pesticide fate and transport (Arnold et al., 1995).

Model Parameters

The drainage area was calculated to be 29.04 ha (71.75 acres). The total area was divided into subbasins determined by management units (greens, fairways, and roughs). The hydrologic routines of SWAT are currently driven by the curve number (CN) methodology. Curve numbers selected for this application were 40 for the greens, 61 for the fairways, and 58 for the rough areas. Precipitation and temperature were measured for the study period.

Irrigation was applied automatically when the soil water dropped below 85% of field capacity. The average slope of the roughs and fairways was approximately 5.5%.

RESULTS AND DISCUSSION

Simulated Surface Water Hydrology

Due to the proximity of the golf course to the airport, run on occurred with precipitation as small as 1.27 mm (0.05 in). It was assumed that no runoff occurred with events less than 6.35 mm (0.25 in). During the 13-month period of record, 17 precipitation events exceeding 6.35 mm (0.25 in) occurred (Table 1). The average daily precipitation amount greater than 6.35 mm was 19.5 mm. Measured runoff ranged from 0.0 to 69.0% of precipitation with an overall average of 22%. The high amount of measured runoff for the two March 1999 events was more likely a result of damming that caused a greater depth and higher discharge than a true runoff amount. The measured data were estimated from open channel flow where control structures were not available.

SWAT simulated runoff for the same time period was considerably lower (Table 1). SWAT underestimated all events except the 17-Oct-98 to 21-Oct-98 event in which the simulated runoff was significantly greater. Model efficiency from all 17 events was calculated at 0.01 (an indication that the model made no significant improvement over the mean). When the two March 1999 events in question were removed the efficiency improved to 0.34. Mean annual simulated irrigation for the greens was 1725 mm while fairway irrigation totaled 1219 mm.

Table 1. Precipitation, measured runoff and SWAT simulated runoff for 17 events from March 1998 to April 1999.

Storm Date	Precipitation (mm)	Measured Runoff (mm)	Simulated Runoff (mm)
30-Mar-98	12.7	2.6	0.0
8-Apr-98	7.1	2.3	0.0
27-May-98	11.7	2.3	0.0
11-Jun-98	13.9	4.4	0.0
28-Jun-98 to 29-Jun-98	9.1	1.8	0.0
23-Aug-98	16.7	0.6	0.0
10-Sep-98 to 12-Sep-98	56.3	10.3	3.3
15-Sep-98 to 16-Sep-98	79.5	13.5	11.8
5-Oct-98 to 6-Oct-98	70.1	16.0	14.3
17-Oct-98 to 21-Oct-98	187.3	39.0	65.3
1-Nov-98	43.2	10.1	3.9
12-Nov-98 to 15-Nov-98	44.8	9.5	0.2
10-Dec-98 to 11-Dec-98	25.8	4.5	0.0
12-Mar-99 to 13-Mar-99	37.6	17.3	0.9
18-Mar-99 to 19-Mar-99	38.1	26.4	2.3
27-Mar-99 to 28-Mar-99	15.0	0.0	0.0
25-Apr-99 to 26-Apr-99	12.5	0.0	0.0
Total	681.4	160.6	102.0

Model Limitations

Initial indications suggests that SWAT is limited in its ability to reproduce the hydrology associated with this turfgrass environment. These limitations may be result of the models current inability to accurately represent the intensive management practices found in turfgrass environments. Limitations with the use of curve number methods for use with event type systems can also explain some discrepancy between measured and simulated values. The CN method was originally designed to give an average curve number for several events to be used over long term periods. The use of curve numbers on individual events was never intended. Thus trying to use curve numbers to reproduce a specific event is difficult and in essence is a misuse of the CN. SWAT is currently being modified to operate in turfgrass environments. Once the modifications are completed the results should improve.

CONCLUSIONS

SWAT was not able to reproduce runoff amounts from specific events. This was attributed to the CN methodology which was designed for long term averaging not event based simulations. Continued turfgrass modifications to SWAT could result in improved hydrologic simulation results. In particular an event based infiltration/runoff model as opposed to CN could improve the excess rainfall predictions.

The simulation model and the simulation strategy demonstrate a potentially significant improvement in methods to evaluate the efficacy of water conservation and water quality protection practices. These strategies can be used on both existing and proposed golf course watersheds. The continued development of SWAT for turfgrass systems will fill a major gap in modeling urban landscapes.

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Modeling and Management of Colbert Hills Golf Course for Water Quality and Ecosystem Diversity[1]

Abstract

Colbert Hills Golf Course, currently being constructed within the Little Kitten Creek Watershed near Manhattan, Kansas, is replacing native prairie grassland and poses concern over its impact on water quality and avian and aquatic species habitat. A watershed water quality model (EUTROMOD) combined with local fish habitation data and Habitat Suitability Index modeling was used to simulate the overall ecosystem impact of projected golf course management actions. Compared to native conditions, good golf course management was simulated to increase annual N yield by 180% and P yield by 24% with little impact on runoff or erosion, while poor management increased N by 630% and P by 24%. This change in aquatic nutrient levels along with concurrent projected changes in stream habitat led to a potential impact on three of the four species analyzed. Avian habitat was also impacted depending upon design and management of non-playing areas. An overall design and management plan was developed with the goal of maintaining predevelopment levels of water quality and habitat. Sustainable water quality management must include proper soil and water conservation, irrigation scheduling, and fertilizer and pesticide management. Appropriate features must be designed into numerous contiguous areas (1.2 ha min. for grassland and 0.8 ha min. for woodland) of non-playing area (rough, riparian areas, etc.) to meet avian habitat needs. And a diversity of stream habitat conditions must be maintained, including clear pool and gravel-substrate riffle zones, and permanent and intermittent streams.

Keywords: Ecosystem health, birds, fish, USLE.

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Introduction

Golf courses have been viewed historically as open-spaces with positive environmental effects. Recently though, golf courses have become associated with heavy pesticide and fertilizer use that may pollute runoff from the course (Watschke et al., 1989). In this light, environmental impacts of construction and management of a new 18+9 hole golf course in northwest Manhattan, Kansas are being examined closely. It is unclear what changes may occur with the conversion of the current prairie ecosystem to a managed golf course ecosystem, including practices such as surface contouring, irrigation, fertilization, pest control, and chemical application. It is important to understand the complex effects the golf course will have on water quality, fish diversity, and native avian habitat because each serves as an indicator of overall environmental quality. The objective of this study was to develop design and management recommendations for Colbert Hills Golf Course to maintain pre-development levels of water quality, aquatic species, and avian species.

Methods

Water Quality

Computer modeling is a common tool used in the development of conservation plans to protect soil, water, wildlife habitat, and many aspects of the environment. By allowing researchers to simulate various conditions within the computer model, management practices can be evaluated and recommendations developed from the results before the practices are implemented. Many computer models are available to simulate a wide array of environmental conditions. EUTROMOD, a watershed and lake modeling eutrophication management tool (Reckhow et al. 1990), was used in this project.

EUTROMOD is a computer spreadsheet that uses land and land use characteristics to estimate runoff, sediment and nutrient yields from a watershed. It uses the Rational Method to estimate runoff and the Universal Soil Loss Equation (USLE) to estimate field erosion. Dissolved and sediment-bound nutrient concentrations are input. Trapping zones are used to account for the portion of erosion that settles out before exiting the watershed.

Colbert Hills Golf Course is located in the Little Kitten Creek watershed outside Manhattan, KS. The watershed receives an average annual rainfall of 76.1 cm. Characteristics used to describe the watershed are shown in Table 1. The 558-ha watershed was divided into three slope classes: steep (>7%, 298 ha), moderate (2-7%, 180 ha), and flat (<2%, 80 ha) based on a 1:24000 scale USGS topographic map. After development, 118 ha of the prairie will be converted to golf course; non-golf-course playing areas remained in prairie conditions (Table 1). The trapping factor was set at 0.72, indicating that 28% of field erosion leaves the watershed each year, based on an assessment of watersheds of similar size and land use in this geo-hydrologic region (Holland, 1971).

Simulated runoff was calibrated for native conditions using the runoff coefficient, and erosion was calibrated by adjusting the cover management practice. The runoff coefficients shown for "non-golf course" areas in Table 1 yielded 191 mm (7.5 in) runoff. This is just 5% higher than runoff (16-yr average streamflow minus an assumed 10% baseflow per Mankin et al., 1999) from a 10.6 km^2 native grass prairie watershed approx. 10 km south of Colbert Hills. Cover management practice of 0.003, representing grass with >95% cover (Wischmeier and Smith, 1978), along with the 72% trapping factor yielded 2.24 Mg/ha (1 ton/ac) sediment delivery, within 1% of a previous study of small basin sediment yields (Holland, 1971).

The EUTROMOD computer model was used to determine and compare runoff volume, sediment yield, and nutrient yield for three land-use situations: native prairie, golf course implemented with poor management techniques, and golf course with good management techniques.

Native Prairie. The native prairie condition was modeled to simulate conditions before golf-course construction began. One of the most important differences between native prairie and golf-course grassland is the application of fertilizers. No fertilizer was applied in the native watershed. The prairie vegetation is less dense than highly manicured turf-grass, but native-grass residues left on the surface would largely offset this difference leading to nearly identical runoff from prairie and golf-course grasses. The dissolved and sediment-attached concentrations of nitrogen (N) and phosphorus (P) for the watershed for the pre-development condition were determined using actual soil and water tests from the watershed (Thien et al., 1998). Nitrogen concentrations for the native prairie were determined to be 1 mg/L dissolved and 3000 mg/kg sediment-attached. Phosphorus at a level of 500 mg/kg was almost entirely in sediment-attached form based on results from Melvern Lake watershed study (Mankin et al., 1999). The results from the native prairie simulation were used as a baseline for comparison to the other simulations.

Table 1. EUTROMOD input parameters.

	Golf Course			Non-Golf Course		
Slope (%)	> 7	2 – 7	< 2	> 7	2 - 7	< 2
Area (ha)	72	43	3	226	137	77
Runoff Coefficient	0.25	0.16	0.10	0.30	0.22	0.14
Rainfall Erosivity [R] (MJ mm ha^{-1} hr^{-1} yr^{-1})	2800	2800	2800	2800	2800	2800
Topographic Length/Slope [LS]	3.60	1.15	0.25	3.60	1.15	0.25
Soil Erodibility [K] (Mg hr MJ^{-1} mm^{-1})	0.32	0.32	0.32	0.32	0.32	0.32
Crop/Cover Management [C]	0.003	0.003	0.003	0.003	0.003	0.003
Conservation Practice [P]	1	1	1	1	1	1

Golf Course with Good Management Practices. A golf course can produce low environmental impact if well managed. Good management techniques include proper application rates of fertilizers and pesticides, effective irrigation with no runoff, and management of rough areas to benefit wildlife and create buffer strips to filter runoff. For the best management situation sediment-attached N was set to the quoted optimum level of 30,000 mg/kg from Keely and Whitney (1998), 10 times that of the native prairie. Dissolved N concentration in runoff remained at 1 mg/L, unchanged from native prairie conditions, assuming that fertilizer applications effectively delivered the nutrient to the soil. No dissolved P was expected to runoff from the golf course, but with an increase of applied P and no increase in runoff volume, the sediment-attached concentration of P was assumed to double from native conditions to 1000 mg/kg. The earthwork and drainage improvements involved in construction of fairways and greens also impacts runoff, which was simulated by improving hydrologic soil group from B to A, which decreased runoff coefficients as shown in Table 1.

Golf Course with Poor Management Practices. Improper golf course management can lead to environmental concern. Sparse vegetation growth can increase the runoff. Improper fertilizer application compounded by poor irrigation timing can lead to increased concentration of N and P in runoff. In addition to the water quality problems, improper management may decrease wildlife-habitat value of rough areas. Dissolved N concentrations for the golf-course runoff were estimated using optimum soil fertilizer application levels (Keely and Whitney, 1998) and a calculation of the runoff volume resulting from a 10-yr, 24-hour storm and a curve number of 61 for the golf course areas; runoff from this type of storm provides a reasonable approximation of one-half the average annual runoff in this region (Mankin et al., 1999). To simulate a worst-case scenario in EUTROMOD, half the recommended application of N was modeled to run off in a 10-yr, 24-hour storm event immediately following fertilizer application, yielding 108 mg/L dissolved N. The sediment-attached N, dissolved P and sediment-attached P were assumed to be the same as with good management practices.

Fish Species

There are five fish species found at the Colbert Hills Golf Course site: Creek Chub (*Semotilus atromaculatus)*, Southern Redbelly Dace (*Phoxinus erythogaster*), Fathead Minnow (*Pimephales promelas*), Central Stoneroller (*Campostoma anomalum*), and Orangethroat Darter (*Estheostoma spectabile*). The fish are all native to Kansas. Table 2 below lists the characteristics needed for survival of each of the fish species.

To determine the levels of N and P that the fish species present in Little Kitten Creek could tolerate, an analysis was completed from data collected by the Kansas Department of Fish and Wildlife, Environmental Services Section (KDWP, 1998). The data contained the numbers and species of fish caught per location, and the level of N and P in the water. Data for these species are fairly representative of regional conditions; the number of sampling locations range from 38 to 56 except for the Southern Redbelly Dace for which only one sampling location was available. The results are summarized in Figure 1.

Survivability of fish species was determined by considering Table 2 and Figure 1. Management factors that create conditions that negatively impact the necessary conditions for a species (Table 2) were assumed to negatively impact their survivability. Also, water quality conditions that fall outside the range each species was proven to tolerate (Figure 1) were assumed to have a similar impact on that species in the Little Kitten Creek. These assumptions represent coarse estimates of water quality and habitat suitability at best, but were considered to be adequate for providing cautionary flags to guide management decisions.

Table 2. Fish Characteristics.

Fish Species	Water Quality	Tolerance	Dietary Needs	Spawning Requirements
Creek Chub	Clear pools of water[†]	Tolerant to turbidity[‡]	Invertebrates and other fish[‡]	Gravel bottoms in flowing water[‡]
Southern Redbelly Dace	Small, clear streams, near source of springs[†]	Moderate tolerance to sedimentation, nutrient enrichment[§]	Algae and occasionally aquatic invertebrates[‡]	Gravel riffles in flowing water[‡]
Fathead Minnow	Intermittent streams with mud or clay bottoms[†]	High tolerance to pollution, warm water, low oxygen[‡]	Microscopic invertebrates, plants, and detritus[‡]	Beneath twigs, stones, and other objects[‡]
Central Stoneroller	Clear streams permanent creeks in Flint Hills[†], clear bottoms[‡]	Tolerant to moderate siltation, turbidity[†]	Algae, encrusted organisms scraped with lower lip[‡]	Gravel pits in flowing water[‡]
Orangethroat Darter	Small streams, shallow riffles with fine gravel or mixed sand bottoms[†]	Tolerant to warm water[†]	Aquatic insects and other small invertebrates[‡]	Bury eggs in gravel[‡]

[†]Fishes in Kansas (Cross and Collins, 1975)
[‡]Fishes of the Central United States (Tomelleri and Eberle, 1990)
[§]Dr. Matt Whiles, Asst. Professor of Entomology, Kansas State University

Avian Habitat

Habitat Suitability Index (HSI) models have been developed to evaluate habitat quality for specific avian species during the spring season. HSI is quantified by combining separate suitability index values that each assess the relative quality of an important habitat component. Bird counts on transects of the golf course site before construction began (1997) indicated an abundance of both grassland and woodland species (Robel and Bye,

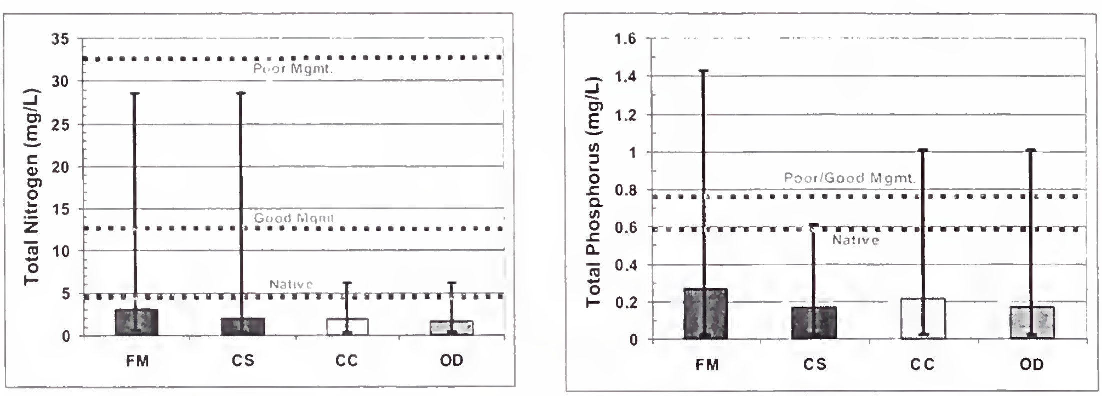

Figure 1. Kansas Department of Wildlife and Parks database total nitrogen and total phosphorous means and ranges for Fathead Minnow (FM), Central Stoneroller (CS), Creek Chub (CC), and Orangethroat Darter (OD). Dotted lines show EUTROMOD simulated output levels for native, good management, and poor management conditions.

unpubl.). To evaluate the overall impact of development, HSI models were selected for two indicator species: Eastern Meadowlark (*Sturnella magna*) to represent grassland species and Field Sparrow (*Spizella pusilla*) to represent woodland species.

Eastern Meadowlarks are omnivores that live in grasslands, meadows, and pastures. Their territory during breeding season can range from 1.2 to 6.1 ha (3 to 15 ac). Important habitat features have been summarized in a HSI model by Schroeder and Sousa (1982), represented by Eq [1]:

$$HSI_{EM} = (V_1 \times V_2 \times V_3 \times V_4)^{1/2} \times V_5 \quad [1]$$

where,

V_1 = ground shaded by projection of non-woody vegetation (%)

V_2 = herbaceous canopy cover that is grass (%)
V_3 = average height from ground to highest tip of herbaceous canopy in Spring (ft)
V_4 = distance to perch site such as forbs, shrubs, tree, fences, or telephones wires (ft)
V_5 = ground shaded by projection of woody shrub canopy less than 5 m height (%)

This equation demonstrates an equal importance of V_1 through V_4, and relatively greater importance of V_5. Optimal habitat quality is obtained when HSI = 1.0.

Field Sparrows prefer old fields with scattered woody vegetation and shrubby or forested areas. Their mean territory size during breeding season is less than 0.8 ha (2 ac). The HSI developed by Sousa (1983) quantifies the key structural habitat features according to Eq [2]:

$$HSI_{FS} = (MIN[V_1, V_2] \times MIN[V_3, V_4])^{1/2} \quad [2]$$

where,

V_1 = ground shaded by projection of woody shrub canopy less than 5 m height (%)
V_2 = total shrubs less than 1.5 m height (%)
V_3 = ground shaded by vertical projection of grass (%)
V_4 = average height from ground to highest tip of herbaceous canopy in Spring (ft)

Habitat suitability is affected by both woody vegetation, via V_1 or V_2, and herbaceous vegetation, through V_3 or V_4. Again, optimal habitat quality is obtained when HSI = 1.0.

Results and Discussion

Water Quality

The pre-development simulation yielded a runoff of 191 mm (7.5 inches), a soil loss of 2.26 Mg/ha-year, and total N and P losses of 4850 kg/year and 630 kg/year, respectively. Using these values average annual concentrations of N and P exiting the watershed were 4.5 mg/L and 0.59 mg/L. Actual in-stream concentrations will vary from this average according to the relative proportions of dissolved and sediment-attached nutrients in each runoff event as well as in baseflow; this may lead to transient high nutrient levels not accounted for in this analysis. These values compare favorably with average in-stream concentrations of 3.9 mg/L N and 0.54 mg/L P estimated for native grass conditions in another Kansas watershed (Mankin et al., 1999).

Good management conditions yielded total annual yields of 12,920 kg or 12.7 mg/L N and 780 kg or 0.77 mg/L P with a slight decrease in annual runoff to 183 mm. With this proper management, the increases in N and P losses from predevelopment conditions are 180% and 24%. The poor management conditions led to the same runoff (183 mm) and soil loss amounts, but increases in total annual yields of N to 33,450 kg or 32.8 mg/L and P to 1330 kg or 0.77 mg/L. This is a 630% increase in N losses and 24% increase in P losses from pre-development conditions.

Fish Species

Four fish species found in Little Kitten Creek were correctly predicted to survive in the pre-development (native) conditions, as indicated by N and P levels within the range of known habitat levels from a Kansas database (Fig. 1). For the post-development with good management condition only the Fathead Minnow will have adequate N and P levels, while for poor management practices no species are within proven ranges of tolerance (Fig. 1). This indicates these species may be challenged in the new environment. Proper fertilizer management is critical.

With the increased amount of soil erosion from the golf course during construction, the turbidity present in the streams will also increase. This will affect the Southern Redbelly Dace, the Creek Chub, and the Central Stoneroller species the most due to their requirement for low turbidity (Table 2). Dietary needs for all the fish species require aquatic invertebrates and insects. Loss of habitat near streams will decrease the food supply. All fish species present in Little Kitten Creek require gravel to spawn. If poor management practices are used on the golf course, erosion will increase leading to sedimentation in the creek destroying valuable spawning habitat.

Avian Habitat

If non-golf course areas are designed and managed for multiple species habitat (i.e., both woodland and grassland species), then tradeoffs are needed. The HSI indices for the two types of species overlap in 3 terms. Optimizing habitat suitability for one species minimizes HSI for the other one. Because of this, there is no single cure-all management system for the non-golf course areas. For example, Figure 2 shows that the best combined habitat for both species is with 20% shrub and 60% grass, yielding an HSI of 0.4 for each Eastern Meadowlark and Field Sparrow. The combined HSI of 0.8 demonstrates a trade-off: even the best attempt to meet the habitat needs of both species yields a lower total HSI than optimizing for one species (HSI = 1.0).

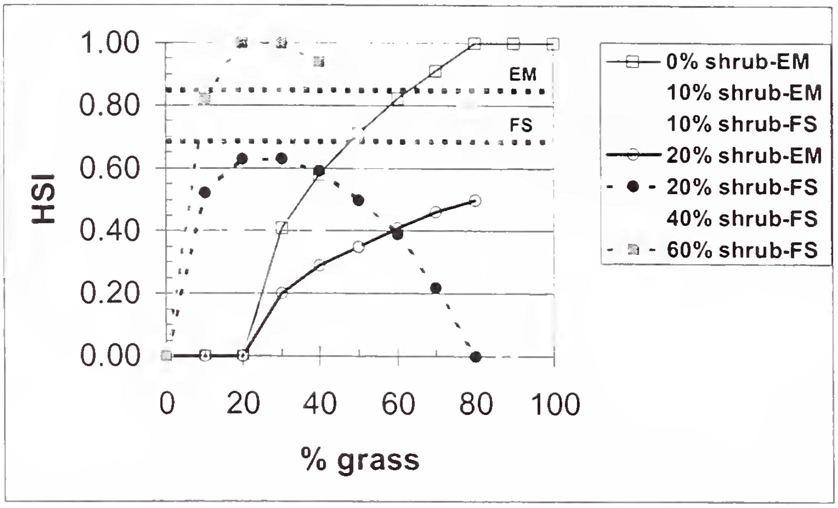

Figure 2. Habitat Suitability Index (HSI) response to % grass and % shrub canopy cover. EM = Eastern Meadowlark. FS = Field Sparrow. Dotted lines represent native habitat HSI values for EM and FS.

It is clear that to maintain both woodland and grassland species in the golf course ecosystem, separate areas will need to be maintained for each. Actual habitat assessments made during bird counts completed in this area prior to construction indicated an Eastern Meadowlark HSI of 0.85 for native grassland and Field Sparrow HSI of 0.68 for native woody draws and riparian habitats (Robel and Bye, unpubl.). These "native conditions" may represent more realistic targets for non-golf course managed areas. An example of conditions that meet native conditions and may be reasonable for golf course "rough" areas are shown in Figure 2. Assumed are values of 90% herbaceous canopy cover, 30 cm average canopy height, 30 m distance to perch site, and 50% shrubs less than 1.5 m, all yielding maximum HSI values for both EM and FS. By varying % grass and % shrub from 0 to 100%, a series of curves were generated for each EM and FS. The figure shows that native HSI values can be attained for FS using 40 to 60% shrub area and 10 to 50% grass area. EM can meet native conditions with 0% shrub area and >60% grass area. Based on minimum species habitat sizes, woodland areas should contain at least 0.8 ha (2 ac) and grassland areas must be at least 1.2 ha (3 ac).

Conclusions

The objective of this project was to develop an integrated design and management plan specific to Colbert Hills Golf Course that will allow maintenance of predevelopment ecosystem health conditions as measured by water quality, avian habitat, and fish diversity. The following components were determined to provide these conditions.

1. Properly schedule fertilizer and irrigation on golf course playing areas to avoid runoff losses.
2. Maintain good grass cover in all areas to minimize erosion and runoff.
3. Include terraces or other breaks in slope on playing areas in steeper topographic zones to minimize erosion and runoff.
4. Design and maintain numerous grassland (rough) areas at least 1.2 ha (3 ac) each with characteristics to optimize eastern meadowlark HSI, as shown in Figure 2.
5. Design and maintain numerous woodland (riparian) areas at least 0.8 ha (2 ac) each with characteristics to optimize field sparrow HSI, as shown in Figure 2.
6. Maintain a diversity of stream habitat conditions including clear pool and gravel-substrate riffle zones, and permanent and intermittent streams.

7. Minimize pesticide use to encourage insect food for bird species, and utilize proper herbicide and pesticide management to minimize runoff losses to streams.

Finally, because little direct data is available to relate golf course design and management to water quality or ecosystem health, careful nutrient and pesticide monitoring in adjacent streams, and periodic sampling of bird and fish populations will provide feedback about the golf course impacts on the ecosystem.

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Influence of Subsurface Anomalies on Vadose Zone Transport [1 2 3]

Abstract

Pesticide leaching to ground water is a potential problem in agricultural production areas of South Georgia where sandy soils are prominent. A procedure to identify and sample preferential pathways in root and vadose zones using non-destructive methods, thereby reducing the expense of water quality sampling, has been established. Detailed investigations were conducted on a 1 ha conventionally managed corn field near Tifton, Georgia. Soil-solution samples collected using suction lysimeters from January 1993 to September 1996 following the application of commercial fertilizer, the herbicide atrazine, and the insecticide carbofuran indicate significant vadose zone transport. Samples indicate high pesticide concentrations (296 ppb of carbofuran) in the vadose zone and significant concentrations in the ground water at 12 m (7 ppb of carbofuran). Because of the difficulty associated with collecting the types of field data necessary to quantify mass transport in the vadose zone, and in particular those related to preferential pathways, a two-dimensional simulation model (VS2DT) was used to further analyze the process. Data and simulation results indicate clay lenses induce ponded water in the vadose zone and impact flow and transport. While point observations of solute concentration of flows to ground water increase in some positions under the clay lens, the composite effect may decrease overall groundwater contamination. Simulations indicate that clay lenses increase the period over which the solutes are delivered to the water table, effectively decreasing the concentration in ground water under the plot at any one time.

Keywords: water quality, pesticides, modeling, preferential transport

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[1] Contribution from the USDA Agricultural Research Service, Southeast Watershed Research Laboratory, PO Box 946, Tifton, GA USA 31793, in cooperation with Univ. of Georgia Coastal Plain Exp. Stn.

[2] All programs and services of the USDA are offered on a nondiscriminatory basis without regard to race, color, national origin, religion, sex, age, marital status, or handicap.

[3] ASAE Paper 992143. David D. Bosch, Research Hydraulic Engineer, Clint C. Truman, Soil Scientist, and Frank M. Davis, Computer Specialist, Southeast Watershed Research Laboratory, USDA Agricultural Research Service, PO Box 946, Tifton, Georgia USA 31793. Contact author (dbosch@tifton.cpes.peachnet.edu).

Introduction

Transport of agricultural chemicals from the root zone results in decreased effectiveness, economic losses, and adverse environmental impacts. Transport of agrichemicals depends on rainfall amount, intensity, and duration; chemical solubility and degradation properties; soil properties; biological processes; and management practices. An additional characteristic which has been found to affect this transport process is soil variability. Studies have shown that macropores and spatial variability within the root zone can dramatically increase the transport rate out of the root zone (Singh and Kanwar, 1991; Birkholzer and Tsang, 1997).

At field and watershed scales, soil properties affecting flow and transport are highly variable in space and time. Layering, aggregation, and macroporosity cause the soil profile to be highly variable. Soil and geologic heterogeneity play a significant role in groundwater contamination. While many studies have focused upon the impact of macropores and preferential flow paths within the root zone, few have studied preferential transport in deeper vadose zone profiles. Horizontally layered areas of reduced conductivity restrict vertical transport and concentrate moisture and solutes, funneling the solute plume. It is hypothesized that preferential pathways in the vadose zone lead to unexpected groundwater contamination in areas where matrix flow, or flow through the bulk soil, would not.

The objectives of this research were to: 1) develop non-destructive methods for interpreting preferential flow pathways within and through root and vadose zones, 2) quantify the concentration of agrichemicals flowing in deep vadose zones containing sub-surface anomalies, 3) develop computer simulations which represent actual field conditions of preferential flow induced by clay lenses, and 4) use the collected data and computer simulations to examine the influence of the preferential pathways on the overall transport rate to the aquifer.

Materials and Methods

Site Description

The site used for this analysis was a 1 ha field located near Plains, Georgia USA. The water-table was approximately 10 m below ground surface. The surface soil is composed of a high permeability loamy sand (Bosch and West, 1998; Shaw, 1998). Soil cores were collected throughout the plot for physical property and hydraulic characterization. The mean sea level elevation of the plot ranges from 132 m on the north side of the plot to 130 m on the south side. The average ground slope is 1.2%.

The field was in a cropping sequence of summer corn (Zea Maize L.) and winter wheat (Triticum aestivum L.) from 1993 through 1995. The corn was planted in March or April of each year. Following harvest in August or early September, wheat was planted as a winter crop in late November to early December. Conventional agricultural management practices were followed. The plot was fertilized each spring and fall. The herbicide atrazine and the insecticide carbofuran were applied separately by a tractor-mounted sprayer each year just prior to planting. The pesticides were surface applied and incorporated to approximately 50 mm.

Identification of Preferential Flow Paths

Site investigations using ground-penetrating radar (GPR) were conducted in 1992 in order to identify subsurface anomalies which could impact agrichemical transport (Truman et al., 1999). GPR provides a continuous, non-destructive profile of the features being investigated (i.e. soil horizons, water tables, clay lenses, geologic strata, etc.) in a relatively short time. A grid was laid out encompassing the entire plot. Subsurface anomalies were identified from GPR charts and hand auger samples were used to verify the soil texture.

While samples were collected throughout the plot, the focus here will be on deep lysimeter samples collected along the southern edge of the plot. Fifteen sites were identified for installation of suction lysimeters within the deep vadose zone. Suction lysimeters were installed along the upper and lower ends of an identified sloping lens (fig. 1) and at equivalent depths for comparison. Groundwater wells were installed throughout the site. The screened intervals were approximately from 9 to 10 m and from 10 to 11 m from the soil surface. Monthly soil-water samples were collected using these suction lysimeters and wells.

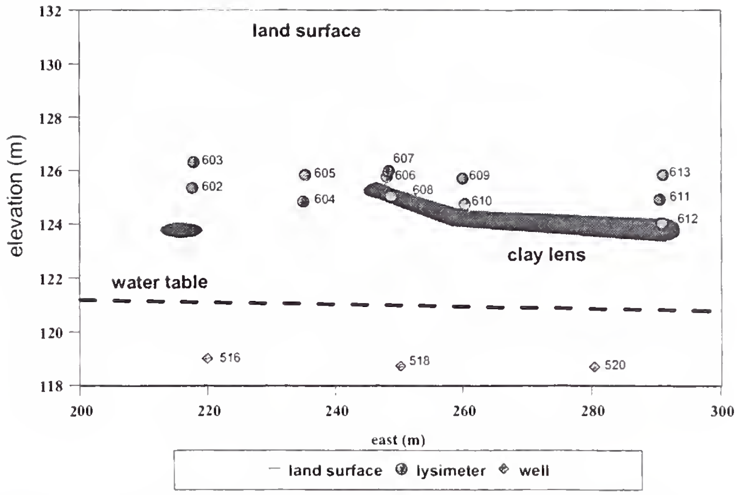

Figure 1. Vertical cross-section of lysimeter locations and wells.

Analytical Methods

All samples were analyzed for nitrate nitrogen (NO_3^- N), chloride (Cl^-), atrazine, and carbofuran. NO_3^- N and Cl^- were analyzed using standard colorimetric methods on a Lachat[4] flow injection system. Ohmicron Corporation enzyme-linked immunosorbent assay (ELISA) methods were used to screen water samples for atrazine and carbofuran concentration. All positive atrazine samples were also analyzed using capillary column gas chromatography (GC).

Numerical Simulation

The Variably Saturated Two Dimensional Transport (VS2DT) simulation model, was used to simulate transport through the vadose zone (Lappala et al., 1987, Healy, 1990). VS2DT is a finite difference program that solves Richards' equation for saturated and unsaturated flow and the advection-dispersion equation for chemical transport. The model was used to gain insight into the transport process, not to make exact comparisons to observed data.

In order to evaluate the effects of the clay lens, two representations were simulated. The first case represented a heterogeneous vadose zone with layering parallel to the soil surface. The profile was divided into 12 horizons with horizon bottoms at 0.25, 0.5, 1.0, 1.25, 1.75, 2.5, 3.0, 3.5, 4.0, 4.5, 5.0, and 10 m. The top layer was a sand, the next three loamy sands, the next three clays, and the final five sands. These layers correspond to the layering found on the western side of the field. The simple case represented the portions of the field without the clay lens. Bosch and West (1998) and Shaw (1998) collected hydraulic conductivity and soil-water release data and parameters were derived for the van Genuchten hydraulic functions (van Genuchten, 1980). Hydraulic and transport parameters are reported in table 1.

To examine the impact of the clay lens, a second simulation grid was established which contained a low density clay lens within the vadose zone extending from x, y coordinates 5 m, 4.5 m to 45 m, 6 m (fig. 2). The slope of the lens was varied to approximate actual conditions. The thickness of the clay lens was assumed to be uniform and equal to 1 m. The hydraulic and transport characteristics of the clay lens are listed in table 1. Aside from the clay lens, the layering was assumed to be the same as in the simple case.

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[4] Trade names and company names are included for the benefit of the reader and do not imply any endorsement or preferential treatment of the products listed by USDA.

Table 1. Hydraulic and transport simulation parameters for the simulated profile.

layer number	saturated hydraulic conductivity ($mmhr^{-1}$)	porosity	residual moisture content	beta	van Genuchten alpha (mm^{-1})	dispersivity (mm)	bulk density ($\mu g\ mm^{-3}$)
1	66.00	0.32	0.05	1.75	0.0039	20	1710
2	48.00	0.37	0.08	2.03	0.0046	40	1670
3	12.10	0.30	0.04	1.60	0.0052	40	1610
4	11.20	0.37	0.08	1.94	0.0042	30	1650
5	4.80	0.32	0.09	1.45	0.0026	70	1690
6	6.30	0.32	0.11	1.61	0.0040	70	1730
7	3.00	0.32	0.10	1.53	0.0033	70	1710
8	35.30	0.37	0.08	2.03	0.0046	20	1710
9	28.40	0.37	0.08	2.03	0.0046	20	1710
10	431.00	0.37	0.08	2.03	0.0046	20	1710
11	40.00	0.37	0.08	2.03	0.0046	20	1710
12	200.00	0.37	0.08	2.03	0.0046	20	1710
clay lens	0.1	0.37	0.1	1.53	0.0033	70	1710

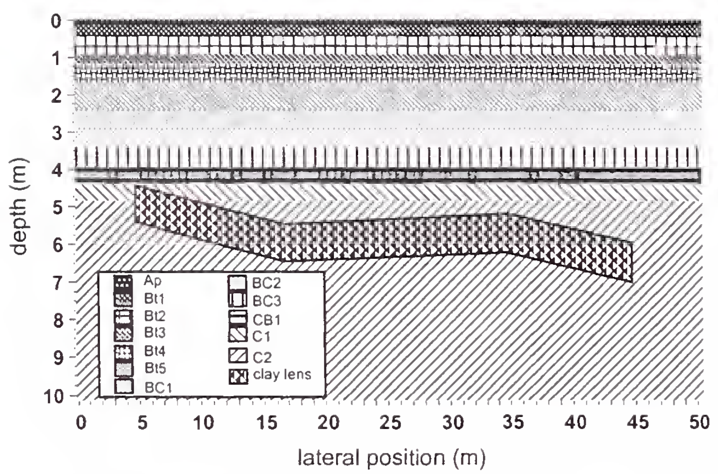

Figure 2. Simulation grid illustrating soil layering.

Both simulation grids were set up to represent a 2-D plane through the plot which was 50 m wide and 10 m to the water table. The horizontal node spacing was set at 1000 mm and the vertical node spacing was set at 31.25 mm. Space and time centering were used. Linear adsorption with Fruendlich n equal to 1 was assumed. Other constants included an anisotropy ratio of 1, specific storage of 1×10^{-7} mm^{-1}, molecular diffusion of 1 mm^2 hr^{-1}, decay constant of 0 hr^{-1}, and an equilibrium constant of 0.

A steady state inflow of precipitation equal to 0.14 mm hr^{-1} at the soil surface was assumed. This approximated the average annual precipitation rate of 1200 mm yr^{-1}. Evapotranspiration at the soil surface was ignored for these simulations. No flow boundary conditions were used for the sides and a constant pressure head equal to 0 was assumed for the water table. For the transport component, a pulse input of a conservative tracer at a concentration of 2009 mg L^{-1} for a period of 80 hrs was simulated. This constitutes a total applied mass of 1125 mg over the 50 m wide by 1 mm unit area, equivalent to an application of 225 kg ha^{-1}. After steady state conditions were established in each grid, the tracer was introduced for the 80 hr duration and then allowed to flow through the grid.

Results and Discussion

Vadose Zone Observations

During installation of the suction lysimeters, zones of saturation were observed just above the clay lens, indicating ponding in these areas due to the low hydraulic conductivities of the lens. A second indication of ponding was the frequency of successful samplings for each lysimeter. Because of the sandy texture, there were times throughout the

course of the study where the soil was not holding enough water to facilitate a sample. The areas directly above the clay lens were frequently saturated and we were seldom not able to capture a sample with these lysimeters. These are indirect indications that the clay lens was restricting vertical flow of water and enhancing lateral flow.

Increases in NO_3^- N and Cl^- in the vadose zone lysimeters were first observed in May 1994, approximately 420 days after first application. The first vadose zone observation of pesticide concentrations above the 0.5 ppb detection limits was made in July 1994 at lysimeters 604 and 609. The following month, atrazine was detected in all lysimeter samples. The greatest observed concentration of carbofuran was 296 ppb, observed at lysimeter 605 which was at 4.3 m, collected in July 1995.

Soil-water samples from lysimeters at 4.3 m indicated elevated concentrations of NO_3^- N, Cl^-, atrazine, and carbofuran along the west side of the main transect. These elevated concentrations appeared to be related to the texture of the surface soil. The texture of the surface soil (top 2 m) was coarser on the west side of the plot than on the east. This appeared to facilitate more rapid transport of agrichemicals on this side. Similar results were found when samples from the lysimeters at 5.2 m were compared.

Groundwater Observations

The first observations of pesticides in ground water at the site were made in April 1994 (fig. 3), approximately 370 days after the first application. The first increases in NO_3^- N and Cl^- were observed in December 1994. The greatest increases in chemical concentration have been observed on the west side of the plot, in well 516. Concentrations up to 75 ppm of NO_3^- N and up to 7 ppb of carbofuran have been observed at this well. Pesticide concentrations peaked in December 1994 while NO_3^- N concentrations did not peak until December 1995. Comparisons of data from wells 516, 518, and 520 indicate elevated NO_3^- N and pesticide concentrations on the west side of the plot. Concentrations also appear to peak later on the west side of the plot.

Vadose Zone Modeling Simulations

VS2DT simulations were used to enhance our interpretation of the vadose zone observations. For the simple case (soil layering only without the clay lens), simulated pressure heads varied from 0 at the water table to -800 mm in layer 12. Simulated moisture contents varied from 15 to 37 %, with the least moisture in layer 12. Since this is essentially a 1-D case, steady state flow rates were constant and equal to the input flow rate, 0.14 mm hr^{-1}. For the simple case, significant levels of the tracer were first simulated at the water table 417 days after application, with all transport to the water table completed by 833 days after application (fig. 4). The mass balance for this simulation accounted for 100.6 % of the introduced tracer out of the grid bottom.

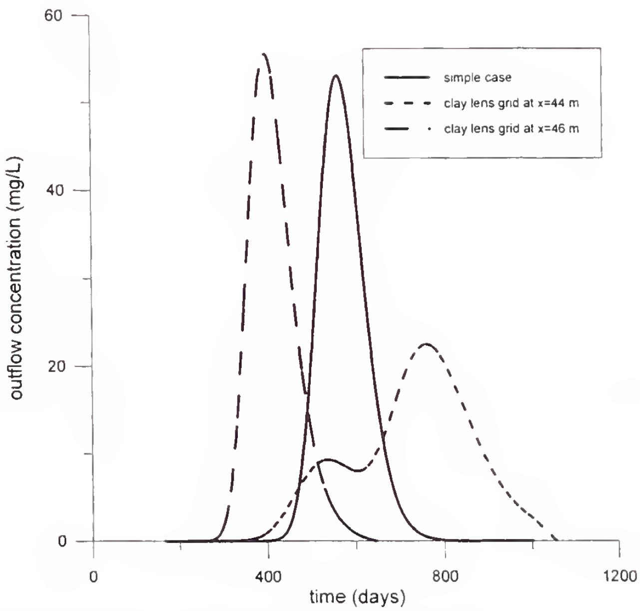

Figure 3. Simulated outflow concentrations for the simple case and the clay lens grid at two positions.

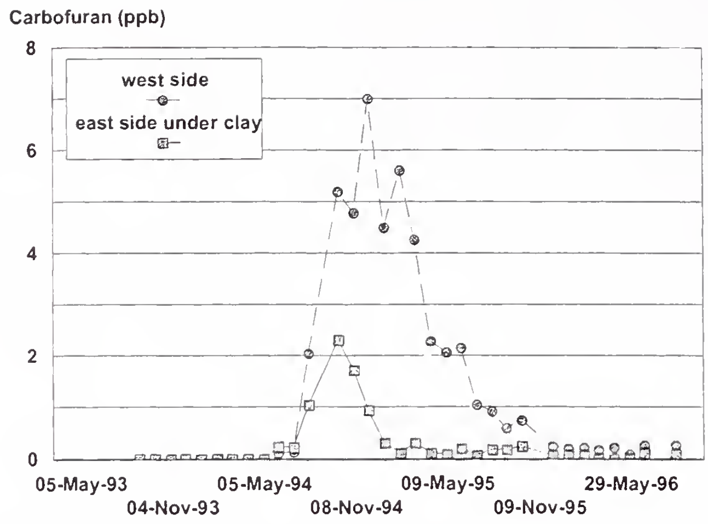

Figure 4. Carbofuran concentrations observed in the plot wells.

The clay lens simulation illustrated several differences to the simple case. Simulated pressure heads varied from 0 to -800 mm, however, ponding was simulated above and saturation within the clay lens. Flow rates ranged from 0 to 126 mm hr^{-1}, with the greatest flows simulated within and near the clay lens. Flow rates were greatest just above the clay lens, where the sandier textured layers become saturated due to the lens. A lateral flow component was induced by the clay lens, with flow driven downslope along the clay.

Significant levels of the conservative tracer were first simulated at 292 days after application (fig. 4). Peak concentrations of 55 mg L^{-1} of the tracer were simulated at 46 m along the x axis. Large concentrations were also simulated from 0 to 4 m, the area not significantly influenced by the clay lens. Loading in this area was similar to the simple case. An interesting phenomenon was simulated at 44 m. At this position two pulses of the tracer were simulated (fig. 4), likely a consequence of flow which by-passed the clay lens on the east side and flow re-directed by the clay lens. This is illustrated somewhat by the histogram of the accumulated mass outflow along the lower boundary (fig. 5). The greatest outflow mass occurs from 45 m to 48 m, followed by the area from 0 to 4 m. There is also significant loading beneath the clay lens, influenced by flow around the lens itself. The total outflow mass accounted for was 1116 mg, or 99% of the input.

Conclusions

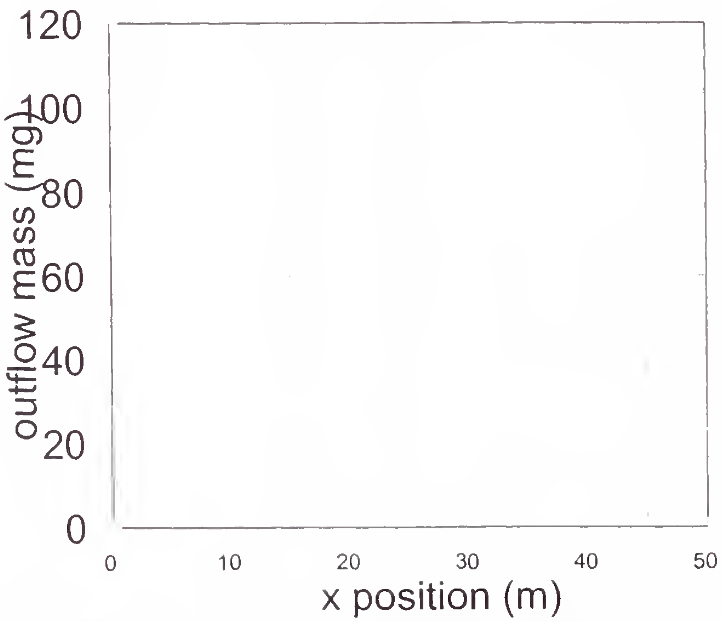

Figure 5. Simulated accumulated mass outflow for the clay lens grid.

The data and computer simulations indicate that clay lenses are impacting flow and transport within the vadose zone. Groundwater data indicate that loading is significantly less under the clay lens (fig. 3). Lower concentrations were observed at the well that was located around 10 m from the end of the clay lens, or at x=35 m on the simulation grid, than at the well located to the west of the clay lens. Interestingly, x=35 m is approximately the point at which the lowest mass loading occurred for the clay lens simulation (fig. 5).

Computer simulations indicate that the lenses can impact groundwater quality, although not in the manner we had expected. Concentrations and mass loadings are affected. The lens channels the flow and transport toward the downgradient end of the lens. However, the lenses do not appear to be having the concentrating effect on the solute transport that we had hypothesized. Because the time at which the solute is delivered to the water table is spread out by the effects of the

lens, the actual concentration in the ground water may be less. While the concentration at individual points may be higher, the concentration over larger areas will be lower because the solute is delivered over greater durations. For the clay lens case, the tracer transport to the water table is spread out over 1,125 days in contrast to a total time of 400 days for the non-lens case. The overall impact thus appears to be to dilute the overall loading to the water table by spreading out the delivery rate.

Our computer simulations appear to match the observed delivery times fairly well. The simulation predicted that the solute would reach the water table earlier on the east side of the plot which was impacted by the clay lens than on the west side which was not impacted. This is supported by our groundwater observations (fig. 3). Approximately 370 to 700 days lapsed between the time of first fertilizer application to the time it was observed at the water table. Our simulations indicate a travel time of 290 to 600 days depending on the configuration with the clay lens.

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Non-Destructive Site Investigation of Preferential Pathways [1, 2]

Abstract

Soils in the Coastal Plain region of Georgia vary in depth, texture, depth to water table, and other chemical, physical, and morphological properties. These soil subsurface features and/or properties influence water and agrichemical (nutrients and pesticides) movement through these soils. Understanding water and agrichemical movement in Coastal Plain soils requires non-destructive methods to determine depth to and lateral extent of subsurface features or properties. Objectives of this paper were to evaluate ground-penetrating radar (GPR) as a tool for 1) identifying and determining the spatial variability of soil subsurface features or properties causing preferential flow of water and agrichemicals and 2) identifying locations which can be instrumented to accurately and efficiently evaluate preferential transport of water and agrichemicals. A Geophysical Survey Systems, Inc. Subsurface Interface Radar (SIR) System-10 impulse radar was used on a 1 ha plot (Troup and Eustis sand) gridded at 10 m intervals. Depth and lateral extent of subsurface features (argillic horizons, clay layers, sand lenses) which have contrasting dielectric properties were accurately measured by GPR. GPR and ground-truth data were used to identify and strategically locate eight sites which were instrumented to evaluate preferential transport of water and agrichemicals. GPR is a valuable research tool because it provides a continuous, non-destructive profile of many subsurface features found in Coastal Plain soils responsible for causing preferential flow in relatively short periods of time.

Keywords: Ground-penetrating radar, agrichemicals

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[2] Trade names have been used for specific information. Their mention does not constitute endorsement by USDA-Agricultural Research Service.

Introduction

Soils developing in the Coastal Plain region of Georgia vary in depth, texture, depth to water table, and other physical and morphological properties. These soil subsurface features or properties influence water and agrichemical movement through these soils. Researchers need improved, efficient, and nondestructive methods to determine lateral extent and depth of diagnostic features found in these soils. Traditional methods of determining soil features or properties involve intense sampling which is time consuming, laborious, and provides limited (point) data that is difficult to relate to surrounding areas or between sampling points. Ground-penetrating radar (GPR) is a research tool that can quickly and accurately provide continuous, nondestructive profiles on depth to and lateral extent of selected soil features, and provide data on soil features between observation points.

Ground-penetrating radar has been used to map soils and geologic materials, detect and determine spatial variability of morphological horizons and water tables, identify preferential flow pathways, and map lake bottoms and provide stage-storage information (Johnson et al., 1979; Bjelm, 1980; Doolittle, 1983; Ryan and McGarity, 1983; Shih and Doolittle, 1984; Olson and Doolittle, 1985; Asmussen et al., 1986; Collins and Doolittle, 1987; Truman et al., 1988; Smith et al., 1989; Vellidis et al., 1989; Steenhuis et al., 1990; Truman et al., 1991).

We are identifying and describing mechanisms and processes controlling water and agrichemical movement in Coastal Plain soils (Bosch et al., 1999). Quality of shallow/perched and deep groundwaters is a major concern, and loss of agrichemicals from root and vadose zones is potentially greatest from soils of this region. These soils are dominated by sandy surfaces with high infiltration rates and conductivities and have subsurface features capable of concentrating water and agrichemicals causing potential groundwater contamination. Information is needed on how agrichemical movement is influenced by spatial variability of soil subsurface features that cause lateral and preferential flow in a non-destructive manner. Objectives of this paper were to evaluate GPR as a tool for 1) identifying and determining depth to and lateral extent of subsurface soil features or properties causing preferential flow of water and agrichemicals and 2) identifying sampling locations which can be instrumented and used to accurately and efficiently evaluate preferential transport of water and agrichemicals.

Materials and Methods

The GPR used in this study was the Geophysical Survey Systems, Inc. SIR System 10 impulse radar. The GPR unit is a broadband, video-pulse radar that provides a nondestructive, continuous profile of subsurface features. The time-scaled system measures the time a short-wave electromagnetic pulse takes to travel from the antenna to a detectable subsurface interface and be reflected back to the antenna. Velocity at which an electromagnetic pulse travels through a material is a function of the effective conductivity and relative dielectric constant of the material. As electrical conductivity increases, the radar signal is dissipated so that maximum probing depth of the radar decreases. Signal quality depends on abrupt changes in dielectric properties. As an interface is detected, reflected signals are intercepted by the radar's receiver. Reflected signals can be displayed and/or recorded. Recorded data was loaded into the RADAN III program for editing or presentation.

All components of the GPR unit were mounted on a small trailer and an antenna with center frequency of 120 MHZ was towed by a small 18-hp tractor at speeds of 0.5-1.0 km h^{-1} (0.3-0.6 mph). Power requirement for the radar system is about 360 watts, and is supplied by three 12-volt, automotive-type batteries connected to a 600-watt generator and battery charger to keep voltage constant.

Site investigations were conducted with GPR to identify subsurface features affecting agrichemical transport. A 1 ha field plot consisting of Troup and Eustis sand was established near Plains, Georgia (Fig. 1). The plot's surface and subsurface morphology was characterized prior to any application of tracers or agrichemicals (Bosch et al., 1999). Selected soil properties with depth and other ground-truth data from inside the plot and border areas surrounding the plot have been measured. GPR was used to extend these properties and depth to dominate subsurface features into the plot. GPR was towed along each transect created by a 10 m by 10 m plot grid. In areas where dominate subsurface features were found, a 5 m by 5 m grid was used. Once initial site investigation was completed, eight sampling locations were identified inside the plot, tracers and agrichemicals were then applied, and then corn (Zea mays L.) was planted. Each sampling location had wells, suction lysimeters, tensiometers, and moisture access tubes at selected depths.

Results

The study area consisted of two major soil types (Fig. 1), the Eustis sand in the NW corner of the plot and the Troup sand in the SE corner of the plot with a transition between the two soils. The Eustis soil has relatively high sand contents, low clay contents, and high K_{sat} values (Table 1). No impermeable soil layer was measured nor

detected with GPR in this soil. However, a sand layer was detected with GPR (Fig. 3). The Troup soil, located in the SE corner of the plot, contained significantly more clay and lower K_{sat} values throughout the profile compared to the Eustis soil.

Table 1. Selected properties with depth for the Troup and Eustis soils.

Soil	Horizon	Depth	Sand	Clay	Ksat
	(cm)	(%)	(%)	(cm/h)	
Troup	Ap	21	85	3	---
	Bt1	92	76	13	12
	BE	52	77	12	19
	Bt4	250	68	27	5
	BC	340	75	22	1
Eustis	Ap	12	91	3	---
	E1	75	90	4	63
	E2	109	89	4	67
	Bt	191	81	15	33
	BC	300	88	14	35
	C	434	92	6	29

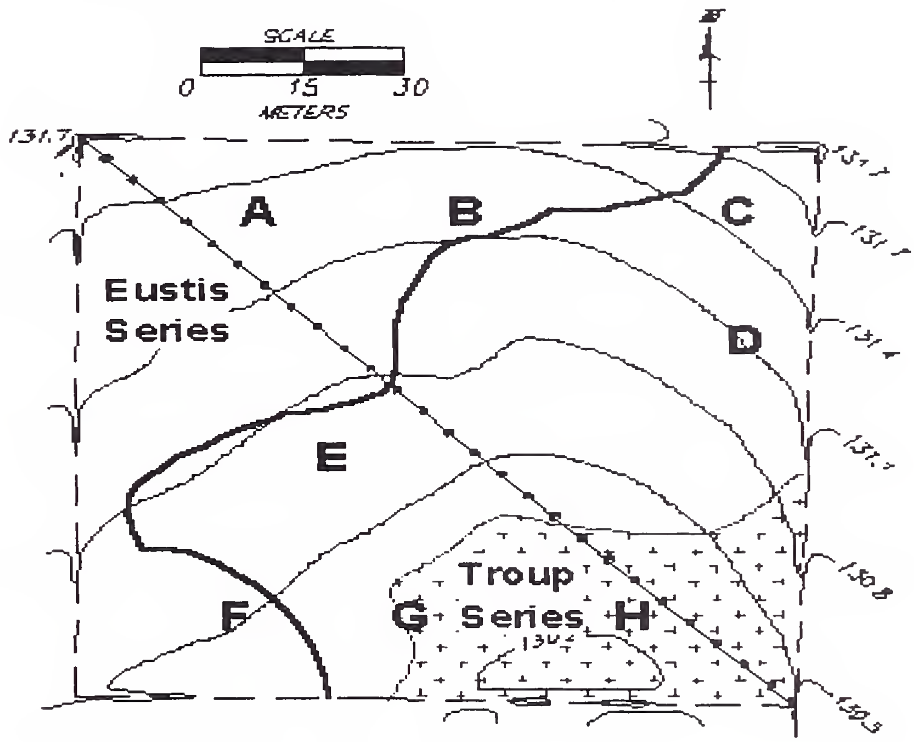

Fig. 1. Surface description of the 1 ha study area (Letters represent sampling sites).

From GPR imagery and ground-truth data, a 3-D map of the entire plot, including the clay and sand layers, and water table was created (Fig. 3). Characterizing field plots in this manner allow for potential preferential flow paths to be identified in a non-destructive manner and sampling locations determined. Knowing the relative position of each of the major soil features of the study area allowed us to strategically place sampling sites (A through H on Fig. 1) to systematically evaluate the overall effect of each soil type/area and feature on water and agrichemical movement. At each site shown in Fig. 1, wells, lysimeters, tensiometers, and moisture access tubes were installed at selected depths. Increased water content and/or saturation occurred at selected times during the year just above the clay layer (at the downslope end), resulting in ponding of water. Indications were that the clay layer restricted vertical water movement thus promoting lateral flow. Additional hydrologic and water quality data from this study area are given by Bosch et al. (1999).

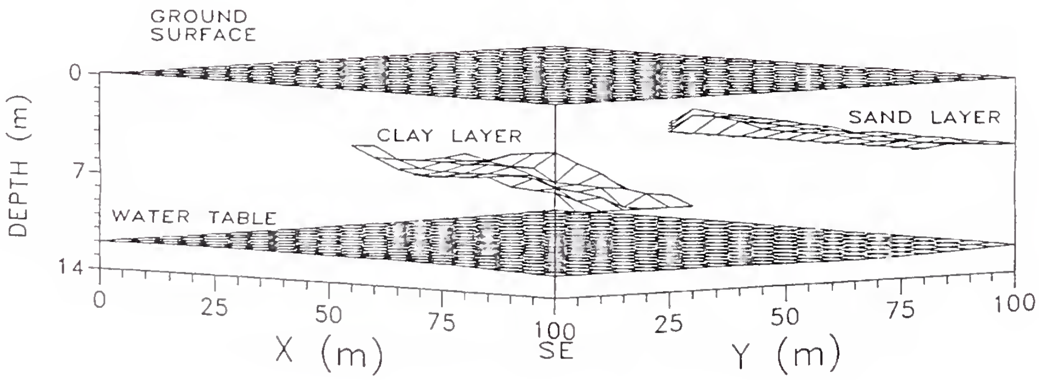

Fig. 3. Three-dimensional image generated from GPR data of 1 ha study area with clay and sand layers.

Conclusions

We used GPR to 1) identify and determine the spatial variability of subsurface soil features or properties causing preferential flow of water and agrichemicals and 2) identify locations which can be instrumented to accurately and efficiently evaluate preferential transport of water and agrichemicals. A Geophysical Survey Systems Inc., Subsurface Interface Radar (SIR) System-10 impulse radar was used on a 1 ha plot (Troup and Eustis sand) gridded at 10 m intervals. The following conclusions can be made:

1. GPR was used to monitor surfaces of subsurface horizons or features which have contrasting dielectric properties.

2. Depth and lateral extent of subsurface features (argillic horizons, water tables, clay and sand layers) which alter or slow water and agrichemical movement were accurately measured by GPR.

3. GPR and ground-truth data were used to identify and stategically locate eight sites which were instrumented to evaluate preferential transport of water and agrichemicals.

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Validation of the Root Zone Water Quality Model (RZWQM): A Review[1]

Abstract

Non-point source (NPS) contaminant transport models have been developed to assess pesticide transport over a large range of topographies, soil types, climatic conditions, and management practices primarily or at least partly because performing site specific field studies is often prohibitively expensive. Today many different models exist and evaluation of these models is necessary to provide information for users to select the most suitable model(s). The purpose of this paper is to summarize and review previous RZWQM water quality evaluations. Other aspects of the model have been evaluated (plant growth, water content and movement, ET), but are not included in this review.

The RZWQM is a physically-based contaminant transport model that includes sub-models to simulate infiltration and runoff, water distribution, and chemical movement in the soil; macropore flow and chemical movement through macropores; evapotranspiration (ET); heat transport; plant growth; organic matter/N cycling; pesticide processes; and chemical transfer to runoff. The effect of management practices on these processes is a centerpoint of the model. To assist the user, RZWQM has: incorporated databases, schemes to estimate soil hydraulic properties from minimum data, initialization wizards, an extensive help system, a Windows 95 user interface, batch mode that allows numerous scenarios to be run, and an analysis option that allows output to be summarized by month or year. The model is usually calibrated using one or two data sets and model evaluation is performed using another independent dataset. The order of calibration was water balance, organic matter pools, and plant growth. The RZWQM adequately simulated NO_3-N in the soil profile and herbicide transport in percolation and runoff in most instances. But some model assessments were unsuccessful in simulating nitrate and pesticide by depth in the soil and NO_3-N in subsurface water and in runoff for which reasons were not investigated in this paper. To accurately simulate herbicide persistence, the two-site equilibrium-kinetic model was utilized, the two-stage dissipation model was utilized, or the herbicide half-life was calibrated.

Keywords: Model validation, Model assessment, Macropore flow

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Introduction

Non-point source (NPS) contaminant transport models have been developed to assess pesticide transport over a large range of topographies, soil types, climatic conditions, and management practices primarily or at least partly because performing site specific field studies is often prohibitively expensive. Today many different models exist and evaluation of these models is necessary to provide information for users to select the most suitable model(s). In light of that, a joint task committee was formed to address "Agricultural Non-point Source Water Quality Models: Their Use and Application". The committee is sponsored by ASCE's Water Resources Engineering Division, Agricultural Water Quality Committee, and the USDA Cooperative State Research, Education , and Extension Service (CSREES) Southern Region Project S-273, "Development and Application of Comprehensive Agricultural Ecosystem Models". As part of this process, numerous models were identified for evaluation. The subject of this paper is the ARS-Root Zone Water Quality Model (RZWQM98). The RZWQM is a physically-based contaminant transport model that includes sub-models to simulate infiltration and runoff, water distribution, and chemical movement in the soil; macropore flow and chemical movement through macropores; evapotranspiration (ET); heat transport; plant growth; organic matter/N cycling; pesticide processes; and chemical transfer to runoff. The effect of management practices on these processes is a centerpoint of the model. To assist the user, RZWQM has: incorporated databases, schemes to estimate soil hydraulic properties from minimum data, initialization wizards, an extensive help system, a Windows 95 user interface, batch mode that allows numerous scenarios to be run, and an analysis option that allows output to be summarized by month or year. The model is usually calibrated using one or two data sets and model evaluation is performed using another independent dataset. The order of calibration was water balance, organic matter pools, and plant growth. It is one-dimensional (vertical into soil profile) and designed for a unit-area basis (point scale). The purpose of this paper is to summarize and review previous RZWQM water quality evaluations. Other aspects of the model have been evaluated (plant growth, water content and movement, ET), but are not included in this review.

Results and Discussion

Nitrate in soil and water

Generally, the RZWQM adequately simulated the NO_3-N content in soil profile (Ma et al., 1998a; Landa et al., 1999; Farahani et al., 1999; Kumar et al., 1998a; Jaynes and Miller, 1999; Cameira et al., 1998). In corn-soybean rotations, the NO_3-N was often over-predicted during corn years and under-predicted during soybean years (Ghidey et al., 1999; Landa et al., 1999; Jaynes and Miller, 1999). When only corn was simulated, the model tended to over-predict NO_3-N (Cameira et al., 1998; Farahani et al., 1999). The over-predicted concentrations may be due to an inadequate simulated temperature response of N cycling in winter (Ma et al., 1998b; Jaynes and Miller, 1999).

The assessments were more mixed on NO_3-N concentration by depth, in subsurface water, and in runoff. Ma et al. (1998b) and Nokes et al. (1996) found the simulated soil NO_3-N concentration at different depths were adequate. But Jaynes and Miller (1999) found that simulated NO_3-N concentration with depth were not adequate and Ma et al. (1998a) found that soil water NO_3-N concentration at 200 cm was under-predicted possibly because of not correctly characterizing the fragipan. A possible weakness of the model is that it over-predicts the peak concentration depth and therefore under-predicts the surface NO_3-N concentration (Ghidey et al., 1999; Jaynes and Miller, 1999; Cameira et al., 1998). The predicted subsurface drainage water or percolate NO_3-N concentrations were found to be reasonable by Ghidey et al. (1999), Kumar et al. (1998a), and Jaynes and Miller (1999) but Farahani et al. (1999) found concentrations were over-predicted on a weekly basis and seasonal basis. Ma et al. (1998a) found simulated NO_3-N runoff concentrations to be comparable to observed but Ghidey et al. (1999) found the runoff concentrations were greatly over-predicted.

Pesticide fate and transport

Generally, the RZWQM adequately simulated herbicide transport in percolation and runoff. When macropores were simulated, Kumar et al. (1998b) and Ghidey et al. (1999) found herbicide movement to subsurface drains or pan lysimeters adequately predicted. On the other hand, Jaynes and Miller (1999) found that pesticide in percolate was adequately modeled without using the macropore option. Runoff was found to be adequately simulated by Ma et al. (1995) when using the two-stage dissipation model and by Ghidey et al. (1999).

As with NO_3-N concentration by depth in soil, the assessments were mixed concerning herbicide concentration by soil depth. Ahuja et al. (1996) and Ma et al. (1995) found RZWQM gave reasonable predictions while Jaynes and

Miller (1999) found that RZWQM did not adequately predict soil pesticide distribution. Jaynes and Miller (1999) observed peak concentrations at the soil surface but the predicted peak concentration was at 15 cm. Both Ahuja et al. (1996) and Azevedo et al. (1997) found the simulated concentrations were generally higher than observed.

Simulated persistence was adequately simulated using the two-site equilibrium-kinetic model (Ahuja et al., 1996) and using the two-stage dissipation model (Ma et al., 1995). In other cases, persistence was either over-predicted (Jaynes and Miller, 1999; Azevedo et al., 1997; and Ghidey et al., 1999) or under-predicted (Wu et al., 1999). Jaynes and Miller (1999) improved the simulated persistence by adjusting the half-life. Therefore, to accurately simulate herbicide persistence, the two-site equilibrium-kinetic model was utilized, the two-stage dissipation model was utilized, or the herbicide half-life was calibrated; using a simple, default half-life resulted in less accurate predictions. It should be noted that although Azevedo et al. (1997) calibrated the half-life, they only considered two values (40 and 60 days) in the calibration.

Conclusions

The RZWQM is a contaminant transport model intended at this time for use by researchers, graduate students, professors, or teachers studying the effects of agricultural management on crop production and environmental quality. Despite the data requirements of a physically based model such as RZWQM, the developers have made it fairly user-friendly. A user can supply available site specific input data and the rest of the input can be calculated by the model, default parameters can be used, or the incorporated databases can be used. Most assessments have found the model to perform adequately with a few exceptions.

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Table 1. RZWQM water quality evaluation summary.

Source	RZWQM components evaluated	Conclusions and comments
Ma et al. (1998b)	Soil NO_3-N concentration (0-30, 30-60, 60-90, 90-120, 120-150 cm);	Generally the calibrated model performed adequately with r^2>0.83 for NO_3-N. The model slightly over-predicted peak concentration possibly because of not properly representing temperature response of N cycling in winter.
Ma et al. (1998a)	Soil NO_3-N in surface 90 cm;	Adequate given the field variability. Over-predicted both control and manured plots at times.
	Soil water NO_3-N at 200 cm;	Over-predicted possibly due to fragipan or unrepresentativeness of the suction lysimeters.
	Runoff NO_3-N;	Comparable to experimental results.
Ahuja et al. (1996)	soil metribuzin and cyanazine concentrations with depth;	Reasonable for metribuzin (r^2=0.83) and cyanazine (r^2=0.97). Somewhat over-predicted both pesticides, especially at lower concentrations.
	cyanazine and metribuzin persistence in soil;	Reasonable using the two-site equilibrium-kinetic model.
Martin and Watts (1999)	soil NO_3-N in 1.5 m soil profile;	Tended to over-predict especially at high application rate possibly because of under-predicted N uptake. The model under-predicted concentration for one early season sampling date (April 19, 1993).
Landa et al. (1999)	soil NO_3-N at 15 cm;	Adequate for corn (tended to over-predict at times). Under-predicted for soybean possibly because of inadequate N fixation rates for soybean and inadequate initialized soil humus pools.
Ghidey et al. (1999)	atrazine and alachlor runoff;	Quite good when predicted and measured runoff agreed.
	NO_3N runoff;	Greatly over-predicted for all events.
	surface herbicide concentration in soil (0-5 cm);	Under estimated early after application. Over-predicted the remaining time. Therefore the model over-predicted the persistence in soil.
	surface (0-5 cm) NO_3-N soil concentrations;	Generally predicted this well but had a tendency to under-predict especially 4 and 8 weeks after application.
	pan lysimeter chemical loss;	Were within the range found in the field (both herbicide and NO_3-N) when macropores were simulated but the field variability was very high.
	NO3-N in soil profile (120 cm) after harvest	Not enough data for analysis, but over-predicted concentration for both corn years, under-predicted for one soybean year, and observed and predicted values were nearly the same for another soybean year.

Table 1. Continued.

Source	RZWQM components evaluated	Conclusions and comments
Nokes et al. (1996)	soil NO_3-N concentration (0-15 and 45-60 cm);	Reasonable
Ma et al. (1995)	atrazine runoff; atrazine persistence; atrazine distribution in soil profile;	Reasonable (R^2=0.92) Reasonable (R^2=0.97) Generally reasonable (R^2=0.73) Note that these results were obtained using the two-stage dissipation model.
Kumar et al. (1998a)	NO_3-N concentration in drainage water; soil NO_3-N concentration in 1.2 m soil profile;	Predicted values followed the measured trends well (R^2=0.88); the annual average NO_3 concentration was within minus 3% of measured values. The model tended to under-predict, especially at higher concentrations. Close agreement (R^2=0.98; slope =0.99)
Kumar et al. (1998b)	atrazine in drainage water;	Reasonable (within ± 12% for two years of data). Note that macropore simulation was necessary to adequately predict annual atrazine losses. Also note that the atrazine half-life and K_{oc} were calibrated.
Jaynes and Miller (1999)	soil NO_3-N concentration with depth; soil NO_3-N in entire soil profile (105 cm); NO_3-N in percolation; metribuzin and atrazine in entire soil profile; metribuzin and atrazine in percolation; metribuzin and atrazine concentration with depth;	Not adequately modeled (observed peaks at soil surface; modeled peaks deeper in soil than observed). Reasonable. Under-predicted early in one season (corn); Over-predicted much of another season (soybean). Reasonable Needed to use a much shorter half-life than published. A two-parameter dissipation model worked better for atrazine but not for metribuzin. Adequate, macropores did not consistently improve predictions. Not adequately modeled. Observed peaks at soil surface; modeled peaks deeper in soil than observed.
Wu et al. (1999)	soil alachlor, metribuzin, and atrazine concentration (surface 15 cm);	Temporal changes were modeled reasonably well but the pesticides were somewhat more persistent than simulated. Simulated concentration had higher peaks and lower tails than oberved.
Farahani et al. (1999)	soil NO_3-N in entire soil profile at end of season; soil NO_3-N by depth at end of season; soil NO_3-N leached;	Reasonable (slightly over-predicted by 2%) Over-predicted above 0.9, under-predicted below 0.9 m. Generally over-predicted on a weekly basis and the seasonal was over-predicted by 79%.

Table 1. Continued.

Source	RZWQM components evaluated	Conclusions and comments
Azevedo et al. (1997)	atrazine concentration with depth;	The predicted concentrations were within an order of magnitude of observed concentrations but the simulated concentrations were generally higher than observed, especially late in the growing season. This may indicate that the persistence was over-predicted. The model correctly predicted depth of atrazine penetration. The predictions may have been improved by simulating: macropores, variation of Koc and half-life, interception by surface residue (faster dissipation on residue).
Cameira et al. (1998)	soil profile (120 cm)NO_3-N;	Reasonable (slightly over-predicted seasonal mean by <5%).
	NO_3-N with depth;	Over-predicted peak concentration depth late in season and tended to under-predict surface concentration. This may be due to over-predicted water movement through a compacted clay layer. The regression analysis indicted that the model tended to over-predict concentration (slope >1).

Simulating the Impacts of Precision Farming and Variable N-Fertilizer Rates on NO_3-N Losses and Crop Yield Using RZWQM[1]

Abstract

The leaching losses of nitrate-nitrogen (NO_3-N) increase greatly as the nitrogen application rate increases than that required for optimum yield. Therefore, the overall objective of the study was to use the RZWQM (Root Zone Water Quality Model; USDA, 1996) to evaluate the effects of N-fertilizer management practices on corn yield and NO_3-N losses with subsurface drainage. The specific objectives of the study were to: (1) calibrate and evaluate the RZWQM using field experimental data for 1996 and 1998 based on subsurface drainage flows, NO_3-N losses, and crop yields under variable N-fertilizer treatments; and (2) determine the optimum fertilizer treatment for the study area based on optimum crop yield and NO_3-N losses with subsurface drainage flows. The field (25 ha in size), located in central Iowa, is drained by nine tile lines (nine plots) with each tile draining into an individual sump. Each sump is equipped with an automatic tile flow recorder which records tile flow continuously on daily basis. Composite water samples were collected weekly from each sump for NO_3-N analysis. Fertilizer (actual N) was applied under randomized block design at the rate of 202 kg/ha in 1996 and 172 kg/ha in 1998 for plots (1,4,9), 135 kg/ha in 1996 and 115 kg/ha in 1998 for plots (2, 5, 8) and 67 kg/ha in 1996 and 57 kg/ha in 1998 for plots (3, 6, 7). Corn was grown in 1996 and 1998. Corn grain yield was measured on nine plots. The hydrologic component of the model i.e. tile flow was calibrated first by using the average values of the soil physical properties measured from the study area. The drainable porosity (difference between porosity and field capacity at 1/3 bar) was the key parameter in controlling the size and shape of the tile flow hydrograph. After calibrating the tile flow simulations, the NO_3-N losses and crop yield simulations were conducted after running the model several times to get the equilibrium conditions for various humus and micro-organism pools. The crop production module of the model was evaluated by comparing the model output for corn grain yield with the measured yield data at different levels of N-fertilizer applications for 1996 and 1998.

Overall, the model predicted tile flow, NO_3-N concentrations, NO_3-N losses, and grain yield satisfactorily by showing an average difference of 6.1%, -6.1%, 0.5%, and 3.8% respectively between predicted and observed values for all the treatments during 1996 and 1998. Model simulations showed that doubling treatment 1 (67 kg/ha) increased grain yield by 41% and NO_3-N losses by 13%, tripling treatment 1 increased grain yield by 68% and NO_3-N losses by 24%, and quadrupling treatment 1 increased grain yield by 61% and NO_3-N losses by 44%. The model simulations showed that the increase in yield was not linear compared with NO_3-N losses as a result of increase in N-application. The yield was found to increase until 200 kg/ha of N-fertilizer was used. After this fertilizer rate yield declined for further increase in fertilizer rate whereas the NO_3-N losses were found higher after this point. Based on the yield and NO_3-N loss analysis, the optimum N-fertilizer treatment (200 kg/ha) was determined but this analysis is very limited and longer duration of yield and climatic data can be more helpful in framing the guidelines for environment safe application of fertilizer inputs while increasing the farmers net returns. Overall analysis of the study shows that RZWQM has the potential to help farmers in making decisions of various N-fertilizer inputs while protecting the soil and water resources.

Keywords: variable N-fertilizer rate, yield, NO_3-N loss, RZWQM simulations

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Introduction

The global nitrate problem is most apparent in the North Central Region of the United States where 83 percent of the nation's corn is produced and 53 percent of the commercial fertilizer is used (Power et al, 1998). The leaching losses of nitrate-nitrogen (NO_3-N) as a result of fertilizer application more than required for optimum yield are further enhanced through the use of subsurface drainage system. The NO_3-N leaching causes environmental degradation on one hand and the economical loss to the farming community on the other hand. Therefore, the overall objective of the study was to use the RZWQM (Root Zone Water Quality Model; USDA, 1996) to evaluate the effects of N-fertilizer management practices on corn yield and NO_3-N losses with subsurface drainage. The specific objectives of the study were to: (1) calibrate and evaluate the RZWQM using field experimental data for 1996 and 1998 based on subsurface drainage flows, NO_3-N losses, and crop yields under variable N-fertilizer treatments; and (2) determine the optimum fertilizer treatment for the study area based on optimum crop yield and NO_3-N losses with subsurface drainage flows.

Water and NO_3-N Transport, Nutrient and Plant Growth Processes in the RZWQM

The RZWQM is a process based model which simulates water, chemical and biological processes to evaluate the impacts of agricultural management practices on water quality through and beyond the root zone. Infiltration into the soil matrix is simulated using the Green-Ampt equation (Ahuja et al, 1995). The NO_3-N transport processes through the soil profile are simulated using a sequential partial displacement and mixing approach in 10 mm soil depth increments based on the concept of miscible displacement. The nutrient processes define carbon and nitrogen transformation processes based on a multi-pool approach for organic matter cycling. The organic matter is distributed over five computational pools and is decomposed by three living micro-organism pools. Process rate equations are based on chemical kinetic theory and are controlled by microbial population size and environmental parameters such as soil temperature, pH, water content and salinity (Hanson et al, 1998). The plant growth model predicts carbon dioxide assimilation, carbon allocation, dark respiration, periodic tissue loss, plant mortality, root growth through the soil profile, water and N uptake, and transpiration. A detailed description of nutrient and plant growth processes can be found from the technical documentation of the RZWQM (USDA-ARS, 1996)

Materials and Methods

Experimental Design and Study Area

The study area is a 25 ha field, owned and managed by a farmer in Central Iowa. The experimental part of the field is drained by nine tile lines with each tile flowing into an individual sump. Each sump is equipped with an automatic tile flow recorder which records tile flow continuously on a daily basis. The composite water samples were collected weekly from each sump for NO_3-N analysis. Three N-fertilizer treatments were applied under a randomized block design each replicated three times. The area drained by one tile line represents one plot (replication) for the treatment. Anhydrous ammonia was injected one week before planting in 1996 and 32% urea ammonium nitrate solution (UAN) was applied in 1998 three weeks after planting corn. Fertilizer (actual N) was applied at the rate of 202 kg/ha in 1996 and 172 kg/ha in 1998 for plots (1,4,9), 135 kg/ha in 1996 and 115 kg/ha in 1998 for plots (2, 5, 8) and 67 kg/ha in 1996 and 57 kg/ha in 1998 for plots (3, 6, 7). Corn was grown in 1996 and 1998. Corn grain yields were measured on nine east west transacts (plots). Primary tillage consists of moldboard plow performed after harvesting the crop whereas secondary tillage consisted of field cultivations, one performed before planting and one during the crop development stage.

Input Data for RZWQM

The minimum data required by the model are daily air temperature (minimum and maximum) and precipitation. On site daily minimum and maximum temperature and wind speed data were available and were used. The hourly rainfall data were available on site and were used to prepare the breakpoint format data for each storm as required by the model. The model requires discretized soil profiles in horizons based on their lithological characteristics. Soil physical properties data of bulk density, porosity, field capacity (1/3 bar), sand, silt, and clay percent were measured at 42 sampling sites in the field and were used as input to the model. The detail of these measurement can be found from Bakhsh et al, (1999)[b] . The management data regarding tillage operations, fertilizer and pesticide applications were available and were used as input to the model. Default values of plant growth parameters were used as recommended in the user's manual (USDA-ARS, 1996).

Model Initialization and Calibration

The initial soil conditions (both physical and chemical) for the soil system of the RZWQM must be input for each specified horizon of the discretized soil profile. Soil moisture content, soil temperature, soil equilibrium chemistry, nutrient chemistry and pesticide concentrations are required as initial input to the model. These initial estimates were adopted from the literature, model user's manual, or the data collected at the experimental site. Initial water table depth was set equal to the tile depth of 1.2 m.

The hydrologic component of the model, i.e. tile flow, was calibrated first by using the average values of the soil physical properties measured from the study area. The lateral saturated hydraulic conductivity of 35 mm/h was used as reported by Singh et al, (1996) and Bakhsh et al, (1999)[a]. The drainable porosity (difference between porosity and field capacity at 1/3 bar) was the key parameter in controlling the size and shape of the tile flow hydrograph. The value of field capacity or porosity was changed until the size and shape of the predicted tile flow hydrograph matched the measured tile flow hydrograph. After calibrating the tile flow simulations, the NO_3-N losses and crop yield simulations were conducted by running the model several times to get the equilibrium conditions for various humus and micro-organism pools.

The organic matter and NO_3-N concentrations data for the discretized soil horizon of the study area were not available. Therefore, the initial values of NO_3-N concentrations, organic matter pools were adopted from the calibrated model for Iowa State University's Northeastern Research Center Nashua, Iowa (Bakhsh et al, 1999)[a]. Several iterations of the model were made under constant management practices of 1996 and the output of every iteration was compared with the measured NO_3-N concentrations in tile flow and corn yield for the study area. This exercise allowed the model to stabilize its interlinked multi-pools by updating the system by reading the initial values from its binary file. The binary file was saved which generated the best matching results and was used during simulations for 1996. The 202 N-fertilizer treatment was used during the iteration process for 1996 which can be considered as the calibration treatment. The impact of the rest of the treatments for 1996 was evaluated by using the calibrated binary file. Whereas the simulations of the model for 1998 were conducted by using stable values of all the pools generated as output using the calibrated binary file.

Results

Table 1 shows the soil physical properties used as input to the model for all plots of the treatments because the treatment mean of cumulative annual drainage volumes were not different at a 5% level of significance. The soil profile depth of 2.52 m was selected to simulate the tile flow component satisfactorily and to accommodate the watertable fluctuations during simulations within the soil profile. The shallow soil horizon was subjected to percolation frequently during simulations rather than giving tile flow. Similar horizons have been reported by Singh et al, (1996) and Bakhsh et al, (1999)[a]. The value of drainable porosity (DP) varied from 0.18 from the bottom layer of the soil horizon to 0.25 to the top layer. The higher value of DP at the top layer is due to higher percent of clay for that horizon. The higher values of bulk density for the bottom horizons were calibrated to reduce the percolation and simulate tile flows.

Table 2 shows the list of site specific calibration parameters. These parameters were calibrated by Bakhsh et al, (1999)[a] for Nashua, Iowa and the same values were used for this site. These regional parameters control the prediction of biomass production. The intra organic matter transfer coefficients are particularly important for N cycling and organic matter degradation. The goodness of fit statistic used was the percentage difference between the observed and predicted indicator variable as mentioned in the user's manual (USDA-ARS, 1996)

Table 3 presents the cumulative predicted and observed tile flows for 1996 and 1998. The N-fertilizer treatment of 202 kg/ha was used during single year iterative runs of the model for 1996. The results obtained during the calibration treatment were well within the 15% range as suggested in the user's manual (USDA-ARS, 1996) for Management System Evaluation Areas (MSEA) sites. The model was evaluated for N-treatments of 135 and 67 kg/ha for 1996 and the variable N-treatments of 1998. Figure 1 compares the observed and predicted tile flows and NO_3-N concentrations in the subsurface drainage water. The model simulations showed that tile flow, NO_3-N concentrations, and grain yield were affected as a result of crop response at different levels of N-fertilizer.

The crop production module of the model was evaluated by comparing the model output for corn grain yield with the measured yield data at different level of N-fertilizer treatments. Figure 2 shows the response of the study area in terms of crop yield and NO_3-N losses with subsurface drainage flows under climatic conditions of 1996 as a result of increase in N-fertilizer rates from one to four times of treatment 1 (67 kg/ha). Model simulations showed that

Table 1. Average measured soil horizon properties of the study area used as input to the model.

Horizon No.	Depth (m)	Bulk Density (Mg/m3)	Porosity	Field Capacity (1/3 bar)	Particle Size Distribution (%)		
					Sand	Silt	Clay
1	0.0-0.43	1.20	0.55	0.30	22	33	45
2	0.43-0.58	1.25	0.52	0.25	21	33	46
3	0.58-0.85	1.30	0.51	0.21	22	32	46
4	0.85-1.15	1.48	0.44	0.19	47	29	24
5	1.15-1.40	1.56	0.41	0.16	35	40	25
6	1.40-1.53	1.75	0.34	0.16	35	40	25
7	1.53-2.52	1.80	0.32	0.14	35	40	25

Table 2. List of crop specific calibration parameters.

Parameters	Corn
Maximum nitrogen uptake rate (g/plant/day)	2.00
Proportion of photosynthesis to respiration	0.12
Amount of biomass needed to obtain leaf area index of 1.0 (g)	10.00
Plant density	60,000
Age effect for propagules as proportion of photosynthesis	0.80
Age effect for seed as proportion of photosynthesis	0.60
Normal maximum root system depth (m)	2.00
Dry mass of the residue on the surface (MT/ha)	4.00
Intra -OM pool transformation coefficients (R14, R23, R34, R43,R45, R53)	0.6, 0.1, 0.1,0.0, 0.3, 0.0

Table 3. Observed and predicted data for the study area.

Year	Rainfall (mm)	Variables	N-fertilizer treatments (actual N in kg/ha in 1996)								
			202			135			67		
			Obs*.	Pred.	%Diff.	Obs*	Pred.	%Diff	Obs*.	Pred.	%Diff
1996	956.7	Tile flow (mm)	253.6[a] (68.7)**	276.2	+8.9	244.5[a] (25.1)**	427.7	+74.9	261.5[a] (28.9)**	291.5	+11.5
		NO_3-N Conc. (mg/L)	21.7 (3.3)**	20.5	-5.5	17.7 (5.2)**	12.0	-32.2	12.6 (2.0)**	15.5	+23.3
		NO_3-N Loss (kg/ha)	61.4	70.6	+14.9	45.6	64.8	+42.1	37.4	47.9	+28.4
		Corn yield (kg/ha)	9796.0[a] (16.3)**	11155.3	+13.9	9829.0[b] (10.4)**	9355.5	-4.8	6639.0[b] (17.8)**	6637.6	-0.03
			N-fertilizer treatments (actual N in kg/ha in 1998)								
			172			115			57		
			Obs*.	Pred.	%Diff.	Obs*	Pred.	%Diff	Obs*.	Pred.	%Diff
1998	796.6	Tile flow (mm)	317.6[a] (76.0)**	220.2	-30.7	304.1[a] (26.4)**	224.7	-26.1	325.9[a] (12.4)**	320.9	-1.6
		NO_3-N Conc. (mg/L)	18.6 (1.4)**	18.2	-2.5	13.6 (0.9)**	12.2	-9.7	10.9 (1.3)**	9.82	-9.9
		NO_3-N Loss (kg/ha)	56.9	37.9	-33.4	41.7	28.6	-31.4	37.2	30.6	-17.6
		Corn yield (kg/ha)	10847.0[a] (12.6)**	11083.2	+2.2	10212.0[b] (9.7)**	10819.2	+5.9	7979.2[c] (12.2)	8442.0	+5.8

* means of treatments; ** standard deviation; means with different letters are different at 5% level of significance; Obs.= observed; Pred.= predicted; %Diff. = percent difference

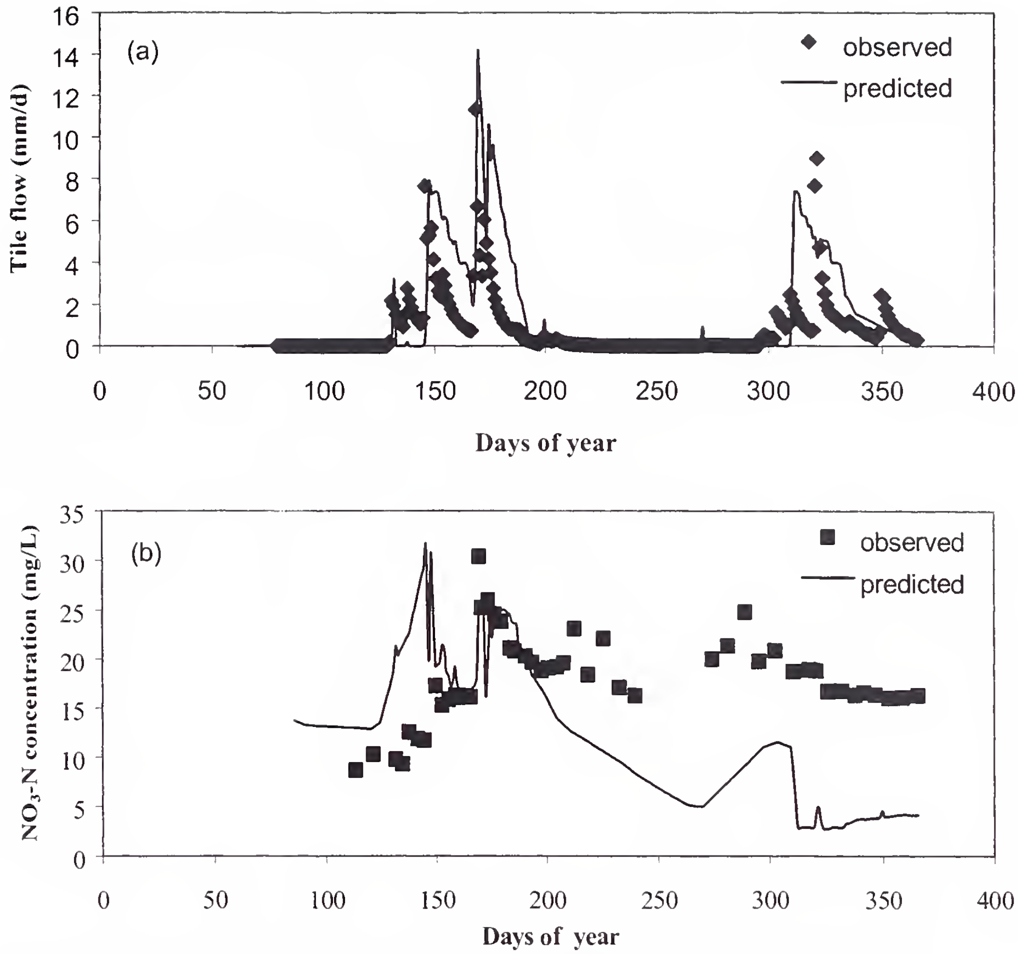

Figure 1.Comparison of observed (average) and predicted data (a) tile flows (b) NO_3-N concentrations in tile flows for 1996 under fertilizer treatment of 135 kg/ha.

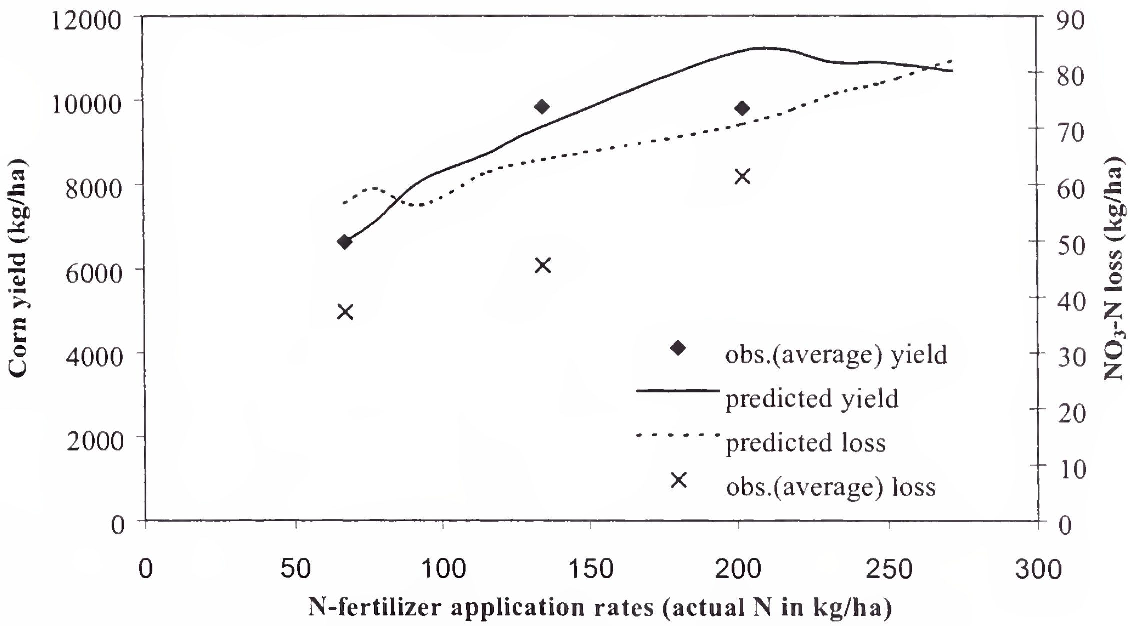

Figure 2. Model simulations for corn yield and NO_3-N losses for different N-application rates for 1996 climatic conditions.

doubling treatment 1 increased grin yield by 41% and NO_3-N losses by 13%, tripling treatment 1 increased grain yield by 68% and NO_3-N losses by 24%, and quadrupling treatment 1 increased grain yield by 61% and NO_3-N losses by 44%. The average difference of predicted yield from the measured yield data for three treatments of 1996 was <5%. The model overestimated NO_3-N losses by 28% (average) for 1996.

Discussion and Conclusions

The model was calibrated and evaluated for 1996 and 1998 when corn was grown during these years. Soybean was grown during 1997. The model was not run for 1997 because separate calibration was required and the data for validation was not available. Figure 1 shows satisfactory matching of the shape of the tile flow hydrograph whereas the shape of tile flow hydrographs for 1998 were different from those of 1996 and the model simulations did not capture the peaks for 1998 as well as for 1996. The simulations (figures) for 1998 are not shown here due to space limitations but soil system response was different for 1998 compared with 1996. The calibration parameters might have been different for 1996 if soybean was grown during 1995. This inconsistency of crops grown during the study period made model calibration critical. This study presents an integrated analysis of soil and water system response to variable N fertilizer treatments. The model simulations for NO_3-N concentrations were found to be under estimated after harvesting the crop. There could be macropore flow effect (not included in this study) which develops during crop development or may be some inappropriate rate factors among soil organic matter pools. The grain yield predictions of the model have been satisfactory throughout. The model simulations showed that the increase in yield was not linear compared with NO_3-N losses for increase in N-fertilizer rates. The yield was found to increase until 200 kg/ha of N-fertilizer was used. After this fertilizer rate, yield declined for further increase in fertilizer whereas the NO_3-N losses were found higher after this point. This analysis shows that the model has the capability to predict the response of soil and water system to different levels of N-fertilizer inputs. Based on model simulations, the following conclusions are drawn:

- Overall, the model predicted tile flow, NO_3-N concentrations, NO_3-N losses satisfactorily by showing an average difference of 6.1%, -6.1%, and 0.5% respectively between predicted and observed values for all the treatments during 1996 and 1998.
- The model predicted grain yield accurately by showing an average difference of 3.8% between predicted and observed data for 1996 and 1998. The yield response curve developed for limited data of 1996 showed that the optimum yield can be obtained with 200 kg/ha of N-fertilizer input.
- This paper presented the calibration and evaluation of RZWQM model for assessing the impact of N-fertilizer treatments on grain yield and NO_3-N losses with subsurface drainage water. Based on the yield and NO_3-N loss analysis, the optimum N-fertilizer treatment was determined but this analysis is very limited and longer duration of yield and climatic data can be more helpful in framing the guidelines for environment safe application of fertilizer inputs while increasing the farmers net returns. Overall analysis of the study shows that RZWQM has the potential to help farmers in making decisions of various N-fertilizer inputs while protecting the soil and water resources.

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Design Guide for Vegetative Filter Strips Using VFSMOD [1]

Abstract

VFSMOD is a field scale, mechanistic, storm-based model designed to route the incoming hydrograph and sedimentograph from an adjacent field through a vegetative filter strip (VFS) and to calculate the outflow, infiltration and sediment trapping efficiency. The model handles time dependent hyetographs, space distributed filter parameters (vegetation roughness or density, slope, infiltration characteristics) and different particle size of the incoming sediment. Any combination of unsteady storm and incoming hydrograph types can be used. The model has been field tested for different soil and climatic conditions in the uplands of the Piedmont and Coastal Plain areas of North Carolina.

A front-end model, UH, was developed to generate the necessary source area inputs for VFSMOD. For each storm, UH generates a rainfall hyetograph, a runoff hydrograph, and sediment loss from the source area. The inputs are generated using a combination of the NRCS curve number method, the unit hydrograph, and the modified Universal Soil Loss Equation based on topography, land use and soil type. With these inputs, VFSMOD simulates overland flow and sediment dynamics within the VFS based on vegetation, soil type, and topography.

The combined models, UH and VFSMOD, are used to begin development of a design guide for representative conditions in the Piedmont region of North Carolina. Simulations were conducted representing a vegetative filter width of 10 m representing a source area to filter width of 8%. Rainfall totals ranging from 16 mm to 150 mm were used to generate 6-hour storm hyetographs and runoff hydrographs from source areas with slopes ranging from 1% to 10%. VFSMOD was used to simulate these conditions using the outputs from UH. Analysis of VFS performance including graphs showing sediment delivery ratios are presented to demonstrate the utility of this approach.

Keywords: runoff, hydrograph, erosion, sediment, vegetative filter strip

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Introduction

Erosion continues to be a major nonpoint source of pollution for surface waters in many parts of the world. Chemicals and pathogens can be transported both in solution and attached to sediment. Phosphorus and some pesticides attached to sediment are a major pollution concern. Several land management practices targeted at the disturbed source area have been suggested to control runoff quantity and quality including conservation tillage contour plowing, and building terraces to reduce the length of slope. In addition, management practices such vegetative filter strips (VFS) have been suggested as potential erosion controls adjacent to the source area. Dillaha et al. (1989) defined these as areas of vegetation designed to remove sediment and other pollutants from surface water runoff by filtration, deposition, and infiltration. Vegetation at the downstream edge of disturbed areas may effectively reduce runoff volume and peak velocity primarily because of the filter's hydraulic roughness, and subsequent augmentation of infiltration. Decreasing flow volume and velocity decreases the transport capacity of the runoff thereby resulting into sediment deposition in the filter. Barfield et al. (1979) reported that grass filter strips have high sediment trapping efficiencies as long as the flow is shallow and uniform, and the filter does not become submerged during the storm event.

Unlike many other land conservation practices, performance and evaluation of vegetative filter strips should be done on a storm by storm basis. Muñoz-Carpena (1993) developed and tested a computer model (VFSMOD) to study hydrology and sediment transport through vegetative filter strips on a storm by storm basis. The model couples a hydrology submodel to describe overland flow and infiltration with a sediment filtration submodel developed at University of Kentucky by Barfield et al. (1979). The resulting model (VFSMOD) can handle complex storm pattern and intensity and varying surface conditions within the vegetative filter strip to evaluate runoff and sediment transport and deposition through the filter. One of the main drawbacks of VFSMOD is that the user must supply inflow hydrographs and sedimentographs from the source area in order to evaluate the filter strip performance.

In an effort to address this limitation to the application of VFSMOD, procedures to generate the source area inputs were implemented in a front-end program (UH). The main objective of the program was to use readily available algorithms and equations to generate inflow hydrographs and sediment for many expected source area conditions. The intent of this effort is to move to the concept of design storms to evaluate vegetative filter strip performance. This paper describes the development of UH to generate the necessary source area inputs for VFSMOD. An example suite of simulations for conditions in the Piedmont of North Carolina are presented to demonstrate how the combined UH-VFSMOD models can be used to evaluate vegetative filter strip performance on a storm by storm basis.

Design Guide Procedures

The procedure for developing a design guide for evaluating VFS performance on a storm by storm basis involves generating the source area inputs for VFSMOD for each design storm of interest. Using the inputs from the source area, VFSMOD simulates the transport and deposition of sediment within the VFS. The inputs from the source area are 1) a runoff hydrograph and 2) the sediment produced from the given storm. These inputs along with a rainfall hyetograph are generated with the model UH using readily available algorithms and equations, which are described in more detail below.

The development of the design guide for a particular area involves determining the size and duration of rainfall events, the range of source area conditions, and the lengths and types of vegetative composition of the VFS of interest. A matrix of inputs is prepared and the UH-VFSMOD models are used to simulate the combinations of inputs. The results are then developed into a graphical presentation to enable easy visual comparison. The resulting presentation can be used to assist users in determining the tradeoffs of management strategies such as increasing VFS length versus implementing improved land conservation practices in the source area.

Hyetograph Generation

Synthetic rainfall hyetographs were generated using equations based on the 24-hr rainfall storms and adjusted to the desired frequency. For storm types II and III, the best-fit approximation can be estimated by (Haan et al., 1994):

$$\frac{P(t)}{P_{24}} = 0.5 + \frac{T}{24}\left(\frac{24.04}{2|T| + 0.04}\right)^{0.75} \qquad [1a]$$

where T=t–12 with t in hours; and P_{24} = the 24 hour total rainfall (cm).

For storm types I and IA, a regression equation was obtained using tabulated data (Haan et al., 1994) as,

(type I):

$$\frac{P}{P_{24}} = \begin{cases} 0.4511 + (t-9.995)(\frac{-0.1617}{-3.0163|t-9.995|+0.013})^{0.5853} & for \quad -3.0163|t-9.995|+0.013<0 \\ 0.5129 & for \quad -3.0163|t-9.995|+0.013>0 \end{cases} \quad [1b]$$

(type IA):

$$\frac{P}{P_{24}} = 0.3919 + (t-7.960)(\frac{0.0843}{120.39|t-7.960|+0.3567})^{0.4228} \quad [1c]$$

with goodness-of-fit parameters of root mean square deviation of 0.0088 and 0.003, and ^{2}=3.363 and 1.539, respectively.

For storm durations less than 24 hours, the ratio of $P(t)/P_{24}$ is used to derive the amount of rainfall at time t from the total rainfall for the period. To generate hyetographs for any duration, D in hours, and storm type, the equation of Haan et al. (1994) was modified to:

$$\frac{P(t)}{P_{24}} = \frac{P(t_{mid}+t-D/2)-P(t_{mid}-D/2)}{P(t_{mid}+D/2)-P(t_{mid}-D/2)} \quad [2]$$

where t_{mid} is 9.995 for storm type I, 7.960 for storm type IA and 12.00 for storm type II and III.

Hydrograph Procedure

The source area hydrograph generation is based on the NRCS (SCS) Curve Number method to determine volume of runoff in a design storm event. This method was developed from many years of storm records for agricultural watersheds in many parts of the United States. The equation is (Haan et al, 1994):

$$Q = \frac{(P-0.2S)^2}{P+0.8S}, for\ P > 0.2S \quad [3]$$

where Q = direct surface runoff depth (mm); P = storm rainfall (mm); S = maximum potential difference between rainfall and runoff (mm). The value of S can be determined by:

$$S = \frac{25400}{CN} - 254 \quad [4]$$

where CN=curve number for the source area. The initial abstraction is assumed to be Ia=0.2 S.

CN is selected antecedent rainfall condition II and can vary from 0 to 100. Antecedent moisture condition II represents moderately wet antecedent moisture conditions. Different land use conditions are used to determine CN and have been tabulated by NRCS (1986). When a combination of different land uses are represented, a composite CN can be calculated as an area-weighted average of the CN for the different land uses.

Using the NRCS method for a triangular hydrograph, the time to peak can be estimated as:

$$t_p = \frac{D}{2} + 0.6t_c = \frac{D}{2} + t_l \quad [5]$$

where t_p = time to peak (hr); D = duration of rainfall (hr); t_c = time of concentration (hr); and t_l = time of lag (hr).

The time of concentration equals $t_l/0.6$ and the longest travel time. This can be determined by:

$$t_c = L^{0.8}\frac{\left[\frac{1000}{CN}-9\right]^{0.7}}{4407S_g^{0.5}} \quad [6]$$

where t_c = time of concentration (hr); L =longest flow length (m); CN= curve number; and S_g = average watershed gradient (m/m).

The TR55 method (USDA NRCS, 1986) is used to calculate the design peak flow:

$$q_p = q_u AQF_p \quad [7]$$

where q_u = unit peak flow (m^3/s.ha.mm) ; A = watershed area (ha); Q = runoff volume (mm); and F_p = ponding factor that accounts for the percentage of the watershed with ponding or wetland condition that will delay the overland flow. The peak unit hydrograph is calculated as,

$$q_u = 4.3046x10^{C_o + C1\log(t_c) + C2(\log(t_c))^2 - 6} \quad [8]$$

where Co, C1, C2 are coefficients obtained for each storm type and the value of the ratio Ia/P.

The source area hydrograph is then calculated from the NRCS unit hydrograph using the approximation (Haan et al., 1994),

$$q = q_p\left(\frac{t}{t_p}e^{1-t/t_p}\right)^{3.77} \quad [9]$$

To couple the generated hydrograph with the storm hyetograph, the hydrograph is delayed t_l seconds. The delay is calculated as the time when initial abstraction (Ia) ends in the watershed, and thus rainfall excess is produced. The delay time is obtained from the hyetograph as the time the rainfall equals Ia.

Modified Universal Soil Loss Equation

Modified Universal Soil Loss Equation or MUSLE is used to compute soil loss for a single storm. MUSLE is a modification of USLE. This equation estimates soil loss from sheet and rill erosion. The equation for MUSLE is (Haan et al., 1994):

$$A = R_m \, K \, LS \, C \, P \qquad [10]$$

where A = computed soil loss per unit area; R_m = storm modified rainfall factor; K = soil erodibility factor; LS = slope length and degree factor; C = crop practice factor; and P = conservation practice factor.

The storm modified rainfall factor (R_m) is the potential of a rainfall event to cause erosion. The storm erosivity due to rainfall is found from the hyetograph by computing EI_{30}. The estimate of R_m for each storm event was done using the equation suggested by Foster et al. (1977) as a combination of the rainfall and runoff factors. The equation is:

$$R_m = 0.5R_{st} + 0.35V(q_p)^{1/3} \qquad [11]$$

where R_{st} is computed as EI_{30}, the rainfall erosivity, and V is the volume of runoff and q_p is the peak runoff rate.

K is the soil erodibility index, which is defined as the mean annual soil loss per unit of erosivity for a standard erosion plot with no conservation, 10% slope and 22 m in length. A smaller K value indicates that the soil is not easily eroded. The length (L) and slope (S) factors are calculated following the original USLE methods. C and P are estimated based on land use and any erosion control practices such as terracing from NRCS tables (see USDA NRCS, 1986).

Vegetative Filter Strip Model

The Vegetative Filter Strip Model (VFSMOD) is a field scale, mechanistic, storm-based model designed to route incoming hydrographs and sedimentographs from an adjacent field through a VFS and calculate the outflow, infiltration and sediment trapping efficiency. The model handles time dependent hyetographs, space distributed filter parameters (vegetation roughness or density, slope, infiltration characteristics) and varying particle sizes of incoming sediment. The model was field tested for soil and climatic conditions in the Piedmont and Coastal Plain of North Carolina (Muñoz-Carpena et al. 1999).

Two main submodels, one for hydrology and one for sediment transport and deposition, are linked together to produce a field-scale single storm model. The model routes the incoming hydrograph and sedimentograph from an adjacent field through a vegetative filter strip (VFS) and calculates the outflow, infiltration and sediment trapping efficiency for that event. Selected inputs for VFSMOD and the UH models are given in Table 1 (Muñoz-Carpena and Parsons, 1999).

Table 1. Input Parameters for UH and VFSMOD

Input Variable	*Description*	*Input Variable*	*Description*
P	amount of storm precipitation	RNA(I)	Manning's roughness for each segment (s.m-1/3)
CN	SCS curve number	SOA(I)	slope at each segment (unit fraction, i.e. no units)
A	area of upstream portion (m^2)	VKS	saturated hydraulic conductivity, K_s (m/s)
storm type	storm type = type (1=I, 2=II, 3=III, 4=Ia)	SAV	Green-Ampt's average suction at wet front(m)
D	storm duration	OS	saturated soil-water content, θ_i (m^3/m^3)
L	length of the source area along the slope (m)	OI	initial soil-water content, θ_s (m^3/m^3)
Y	slope of the source area (%)	SM	maximum surface storage (m)
FWIDTH	width of the strip (m)	SS	spacing of the filter media elements (cm)
VL	length of the plane (m)	VN	filter media (grass) Manning's n m (0.012 for cylindrical media) (s.cm-1/3)
NPROP	number of segments with different surface properties (slope or roughness)	H	filter media height (cm)
SX(I)	X distance from the beginning on the filter, in which the segment of uniform surface properties ends (m).	VN2	bare surface Manning's n for sediment inundated area and overland flow (s.m-1/3)

Example of the Design Procedure for North Carolina Piedmont Conditions

The combination of UH and VFSMOD was applied to North Carolina (NC) Piedmont conditions. Input variables such as rainfall amount and slope were selected to simulate conditions representative of the region. Table 2 shows the variables and their range for this study. The rainfall amounts were selected based on totals for 6-hour storms. For the NC piedmont, a 25

mm storm is fairly typical whereas a 6-hour storm total of 100 mm would have a longer return period (on the order of 10 years). Curve numbers of 78 and 85 were selected for the source area to represent moderate and higher erosion. The slope of the vegetative filter and the source area ranged from 1% to 10% with a filter length of 10 m and a ratio of filter to source area of 8%. The density of the grass in the filter was assumed to be dense (roughness coefficient of 0.45). Clay and clay loam were used to represent NC Piedmont conditions. Simulations were conducted with all combinations of the inputs to demonstrate the approach to develop relationships indicative of vegetative filter conditions over this range of conditions. It should be noted that these simulations represent a small portion of the range of conditions required for a complete design guide. These are intended to demonstrate the capabilities of the combined UH and VFSMOD models.

Table 2. Inputs Used to Simulate Conditions for the NC Piedmont

6-hour Storm Rainfall Amounts (mm)	*Curve Number*	*Slopes (%)*	*Soil Type*	*Filter Density (roughness coeff)*	*Buffer Length (m)*
16, 25, 50	78	1, 2, 4	Clay	Dense (0.45)	10
75, 100, 150	85	6, 8, 10	Clay loam		

Summary outputs from VFSMOD were used to examine the effectiveness of the vegetative filter. Comparisons of runoff inflow and outflow indicated that in most cases there was little difference for the clay and clay loam soils. This is due in part to the low infiltration rate in these soils. An example is given in Figures 1 (simulated hyetograph) and 2 (simulated runoff inflow and outflow) for CN=78, Rainfall=25 mm, Slope=4% and a buffer length of 10 m.

The primary function of the vegetative filters is the reduction of sediment. The sediment delivery ratio is defined as SDR = Sediment Out/Sediment In. This was computed for each simulation. An SDR near zero indicates that the filter trapped nearly all of the sediment whereas values approaching one indicate poor filter sediment trapping.

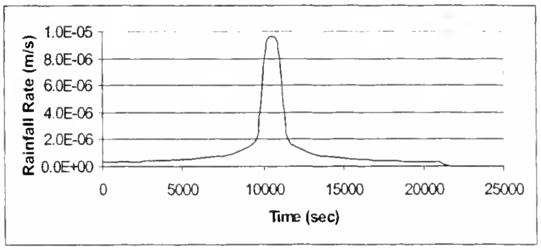

Figure 1. Simulated Hyetograph for the 6-hour Storm with Total Rainfall=25 mm.

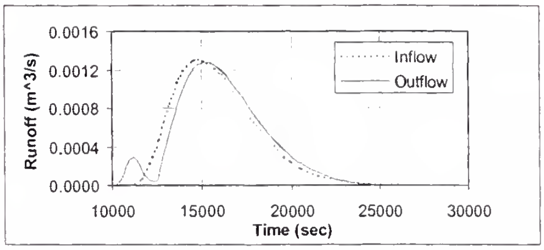

Figure 2. Runoff Hydrographs for CN=78, Rainfall=25 mm, Slope=4% and a buffer length of 10 m.

Table 2 shows the SDR along with the sediment mass in and out of the vegetative filter. For all simulations, the filter strips were effective for the 16 mm rainfall storm indicating that the 10 m buffer length would trap all sediment from this frequent storm size. The increase of the storm size to 50 mm indicated that the SDR increased to 0.49 or greater. To maintain high sediment trapping, an increase in filter strip length or better source area conservation would be required. In general, Table 2 shows that the SDR for the 10-m filter width increases with increases in rainfall and slope as would be expected.

Table 2. Sediment Input and Output (kg/ha) and Sediment Delivery Ratio (SDR) for Selected Simulations.

Rainfall vs. slope (CN=78)									
Slope (%)	*16 mm*			*50 mm*			*150 mm*		
	Input	*Output*	*SDR*	*Input*	*Output*	*SDR*	*Input*	*Output*	*SDR*
4	69	0	0	1129	549	0.49	13337	12544	0.94
10	297	0	0	4954	3992	0.81	58558	57452	0.98
Rainfall vs. slope (CN=85)									
Slope	*16 mm*			*50 mm*			*150 mm*		
	Input	*Output*	*SDR*	*Input*	*Output*	*SDR*	*Input*	*Output*	*SDR*
4	77	0.5	0.01	1453	899	0.62	14719	14028	0.95
10	338	28	0.08	6436	5569	0.87	64814	63773	0.98
Rainfall vs. CN (Slope=4%)									
CN	*16 mm*			*50 mm*			*150 mm*		
	Input	*Output*	*SDR*	*Input*	*Output*	*SDR*	*Input*	*Output*	*SDR*
78	69	0	0	1129	549	0.49	13337	12544	0.94
85	77	0.5	0.01	1453	899	0.62	14719	140278	0.95
Rainfall vs. soil type (slope 4%, CN=78)									
Soil type	*16 mm*			*50 mm*			*150 mm*		
	Input	*Output*	*SDR*	*Input*	*Output*	*SDR*	*Input*	*Output*	*SDR*
Clay	69	0	0	1129	549	0.49	13337	12544	0.94
Clay loam	90	0	0	1469	857	0.58	17409	16695	0.96

Figures 3 and 4 show the SDRs simulated for source areas with CN=78 and 85, respectively. In both cases, the simulations for source and filter areas with a slope of 1% were unable to handle 6-hour storms with rainfall amounts greater than 50 mm. This was due to the vegetative filters filling with sediment prior to the end of storm. For slopes greater than 1%, the SDR increased as 6-hour storm rainfall increased. For a slope of 2%, the 10-m vegetative filter strips reduced sediment outflow by nearly 80% (SDR=0.2) for a rainfall amount of 50-mm and a source area CN=78. For the 50 mm rainfall amount, SDR increased from less than 0.1 (S=1%) to about 0.8 (S=10%) indicating that better source area conservation practices are required with increasing slopes (Figure 3). As one would expect, increasing the CN from 78 to 85, increased SDR for each rainfall amount (Figure 4). Again, this indicates that better conservation practices in the source area would be necessary to keep SDR low.

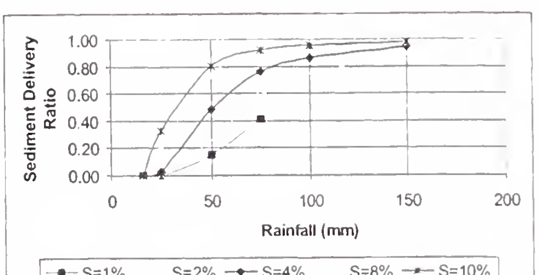

Figure 3. Comparison of SDR for Varying Source Area Slopes for 6-hour Storms (CN=78)

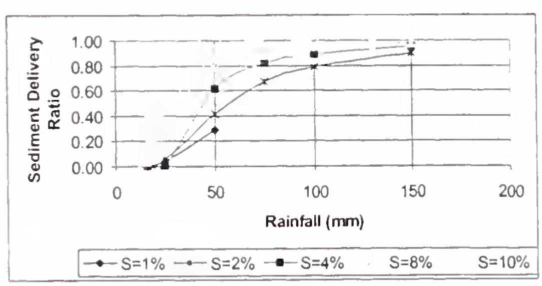

Figure 4. Comparison of SDR for Varying Source Area Slopes for 6-hour Storms (CN=85)

Summary

Simulations using the combined models UH and VFSMOD were done to represent conditions similar to that of North Carolina Piedmont area. The UH model generates rainfall hyetographs, runoff hydrographs and storm estimates of sediment loss from the source area using the NRCS curve number method, unit hydrograph method, and modified Universal Soil Loss Equation. These inputs are used by VFSMOD to simulate overland flow and sediment transport, based on vegetation, soil type, and topography in the vegetative filter strip. Rainfall and topographic factors of both the source area and the vegetative filter strip can be varied to simulate a wide range of conditions and model outputs enable analysis of VFS performance.

Examples representing conditions in the NC Piedmont were done using the UH and VFSMOD. A vegetative filter length of 10 m representing a ratio of source area to filter strip of 8% was used. Runoff hydrographs and storm based erosion losses

using 6 hour storms ranging from 16-mm to 150-mm were simulated with source area CNs of 78 and 85. SDR results were analyzed to demonstrate the use of these models to evaluate vegetative filter strip performance on a storm event basis.

Future work with the combined models will include the development of a full procedure to utilize these models to evaluate the effectiveness of vegetative filter strips for erosion control in a variety of landscape settings.

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Evolution of Instructional Approaches and Materials for Teaching SWAT to Undergraduates[1]

Abstract

With the development and acceptance of erosion and water quality models (EWQMs), it was proposed that instruction of these models be introduced at the undergraduate level. Two of the objectives of the project are to: (1) develop undergraduate EWQM instructional material; and (2) train A&T undergraduates in EWQM. To achieve these two objectives, the undergraduate students would attend EWQM workshops, attend an EWQM class that was developed as part of this project, and develop instructional materials to be used in class and on the Internet. This paper discusses the evolution of the instructional approaches and materials for EWQMs and SWAT.

We concluded that in order for undergraduate students to obtain a grasp of SWAT, they first need to have a good physical "hands-on" understanding of the watershed, subbasin areas, reservoirs, and chemical processes. Our EWQM students constructed 2-dimensional and 3-dimensional physical models of the watershed, a subbasin, and a reservoir. Furthermore, a plexi-glass model representing a cross-section of a subbasin was built. These are used for the initial classroom instruction of delineating watersheds, subbasins, calculating reservoir volumes, showing the different processes of chemical movement, and the concepts of water flow within the soil profile. Classroom team assignments of depicting specific parameters of SWAT using physical models will be assigned. The students are then instructed on the actual running of SWAT. This incorporates the *Internet based instructional material*, corresponding SWAT exercises, and general help sections. They should now have a basic grasp of the use and functionality of the model, and classroom theoretical concepts can now be emphasized. With this order of instruction, from broad to refined, the undergraduate student should successfully acquire a basic understanding of SWAT.

Keywords

SWAT, Internet, instructional, undergraduate, EWQM

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[1] Paper No. 992148 – Previously titled: "An Internet Based Tutorial on the Soil and Water Assessment Tool Model", Peggy Fersner, PE, Adjunct Assistant Professor, Dr. Manuel Reyes, Associate Professor, and L. Waters, A. Grubbs, E. Phillips, B. Mitchell, and C. Lynch, undergraduates, Agricultural and Biosystems Engineering, North Carolina A&T State University, Greensboro, NC 27411, fersner@ncat.edu and reyes@ncat.edu Funding was provided by USDA, CSREES Capacity Building Grants Program.

Introduction

A variety of erosion and water quality models (EWQMs) have been developed in conjunction with the United States Department of Agriculture (USDA), universities and other federal agencies. These EWQMs are becoming standard tools for natural resource management. With this in mind, it was proposed that EWQM instruction would be introduced at the undergraduate level.

This portion is a component of the USDA funded project titled "Strengthening A&T's Instructional Capabilities in Erosion and Water Quality Modeling". Two of the objectives of the project are: (1) to develop undergraduate EWQM instructional materials; and (2) to train A&T undergraduates in EWQM. To achieve these two objectives, the undergraduate students would attend EWQM workshops, attend an EWQM class that was developed as part of this project, and develop instructional materials to be used in class and on the Internet. Like all good things that involve undergraduate students, this turned out to be a trial and error project.

The undergraduate students of today definitely understand the term "plug and play". They have been weaned on graphical user interfaces (GUIs) in computer technology. The Soil and Water Assessment Tool (SWAT), therefore, is a very familiar interface. By doing some right and left clicking of the mouse, they are easily able to explore the model. Intimidation is not a word that they know. The "help" areas of SWAT are complete in the way they give the definitions and the range of values. However, here lies the problem. As undergraduates, some of the parameters may be familiar to them from previous courses, while the majority of them will not mean a thing. Some of the parameters will be out of the scope of undergraduate understanding. With this in mind, we set out to involve the students in the development of instructional materials for a variety of EWQMs. This paper will discuss primarily the evolution of the instructional approaches and materials for SWAT.

Methodology

What Didn't Work

Initially, it was setup so that interested and selected students would be given a stipend for each semester of duties that they fulfilled. These were spelled out in detail and the students were required to sign the agreement. The requirements were broken into five basic areas.

- Academic Policy
- Work Schedule
- Travel
- Instructional Material
- Payment

Academically, the students were required to have a 3.0 GPA, attend an EWQM class, and attend a workshop pertinent to the particular EWQM that they would be studying. The first two items were accomplished easily. The attendance at the workshops took some arm-twisting and agreement waving since they were routinely offered at the end of semesters or over breaks. For those students that attended the workshops, they learned the process of running that particular EWQM. They usually did not pick up any knowledge as to how to select or arrive at values for different parameters.

The students were required to set a definitive work schedule each semester that would entail ten (10) hours per week with any time changes needing a faculty member's approval. Each week, the students and their supervising faculty member would meet at a set time to establish the tasks to be accomplished, to examine long term goals, and to review the progress made during the previous week. With this area we found:

- The majority of students needed extremely detailed instructions for their assignments.
- As faculty, we never had a good grasp if they were actually working during their scheduled 10 hours. They may have been in the building, but e-mail and assignments tended to take priority.
- The success of this method worked only as well as we could supervise the individual student. It became time prohibitive.

Next, we tried adding a group meeting once a week. We hoped that this group meeting would keep the students current on their assignments and perhaps instruct other students as to what had been successful in the search for information or what had not. This proved to be a little more successful, but the work was still not getting done at a rate we considered acceptable. It was definitely time to rethink our strategies.

The instructional material was to be a derivative of the weekly assignments. We had hoped that once students had researched an area, such as explaining the delineation of a watershed, they could develop a series of exercises that would walk other students through the process. That never became a reality.

Summary of What Didn't Work

- With outside work schedules, 16-18 credit hours and then 10 hours of EWQM time, the EWQM work was what got pushed aside.
- Success of the assignments depended on the motivation of the student, not so much on grade level or their GPA.
- It became easier to do the research ourselves since we were practically walking the students through the process step-by-step.
- The students did not have a good grasp of developing instructional material.

Starting to Rethink the Methodology

It became apparent that the students just didn't have the time or motivation outside of class with competing activities to complete this work. Also, anything that seemed too much like a written homework assignment was not well received. Therefore, in addition to a stipend, cash awards would be given for projects that were treated as major assignments given within the EWQM class. The class was divided into teams for each project. The first project was the development of instructional material for WEPP. They were asked to:

- Present runoff and erosion rates for the Environmental Studies Lab as it exists
- Present runoff and erosion rates for the same area after corn, soybean and sorghum had been planted.

They were to include in detail how they arrived at each WEPP parameter and the steps to get that parameter. Again, these were to be used for future instructional material. This was moderately successful. The weak spot was again the instructional exercises. We did discover that the students enjoyed ***constructing*** educational aids. The focus then switched to that area. The next project was to develop a three-dimensional model that would explain certain parameters used in DRAINMOD. Once that was accomplished, the teams were instructed to show the class how these parameters would be input into DRAINMOD. A few of the required parameters were:

- Drain depth
- Spacing between drains
- Water table height between drains
- Hydraulic conductivity
- Capillary rise

This project created quite a bit of serious competition with some very creative ideas. The weakest portion was, as we have now learned to predict, the instructional portion of input into the EWQM.

Figure 1 – Shannon Adams and Diane Waters with their model

Summary of What We Learned with this Approach

- Students will work very hard within the confines of a classroom assignment for cash rewards and a grade
- Students enjoy the process of constructing physical models

Getting it Right

Third times the charm or so they say. Once more we stepped back and looked at how we best could utilize the undergraduate students in creating instructional material. It had become obvious that they really enjoyed constructing physical models. We looked at what we perceived to be common elements among the majority of the EWQMs and where physical models could be included. We recognized that students learn differently and some have a difficult time switching from 3-dimensional to 2-dimensional thinking. This had become obvious every time a watershed or subbasin had to be delineated. It became apparent that we would use the students to construct 2-dimensional and 3-dimensional models for use with the EWQMs. This proved to be very successful.

In spite of our trial and error methods, all of our EWQM students have stayed in the Agricultural and Biosystems Engineering program. We feel that the personal involvement of the student in a project and the one-on-one time with a faculty member has definitely helped ABE retention. We do recognize that the stipend was also a good incentive.

Figure 2 – Students with 3-dimensional model of the Environmental Studies Laboratory under construction (L-R) Anthony Grubbs, Eric Phillips, Bryan Mitchell and Diane Waters

Development of Internet Instructional Material

In trying to find unifying instructional areas among the EWQMs, we were led to three main subject areas.

- General overview of the model
- Basic parameterization of the site – use elements common to the majority of EWQMs
- Interpreting outputs

The general overview of the model and interpreting the output, would be specific for each EWQM. Some overlap would occur on basic parameters such as climate or soils. These exercises to be used for multiple EWQMs. Instructions for input of these parameters would vary. With this in mind, a series of instructional materials and basic exercises were developed for use in teaching SWAT.

1. General overview of what the model can do
2. Parameterization of the site
   a. Climate – how do we get it from a database and then how do we input it
   b. Soil – where do we find this information and what do we do with it
   c. Topographic characteristics – determination of the:

- delineation a watershed and subbasins
- average channel slope
- average channel length
- average channel width
- average watershed slope
- average watershed length
- Reservoir characteristics
- Management

3. Interpreting Outputs

Setup of the Internet Site

The EWQM site will be accessible from the Internet. A homepage for erosion and water quality models is established at http://152.8.7.153/ewqm. From there, it is a simple matter of clicking on the particular EWQM that is of interest. The hierarchy of the site is shown below.

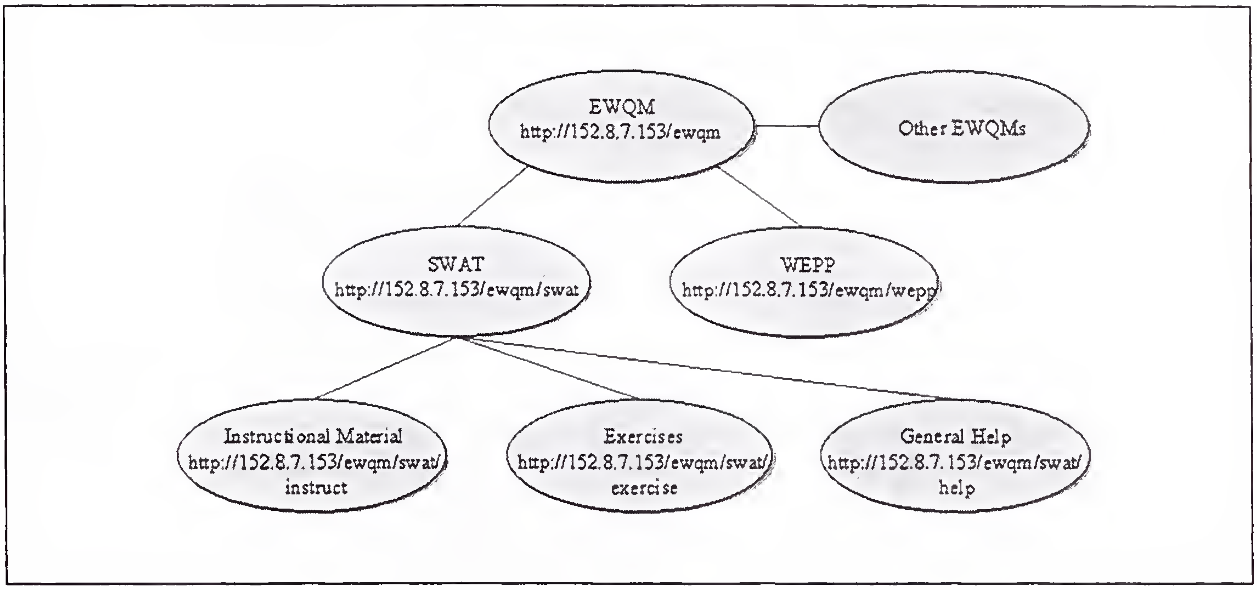

Figure 3 – Schematic of EWQM Web Site

Instructional and Exercise Material

Each specific EWQM will be broken into at least three general areas; 1) instructional material, 2) exercises, and 3) general help. The instructional material will concentrate on those areas that we feel undergraduates will be able to grasp and that will benefit their overall understanding and appreciation of EWQMs. The exercises will parallel the instructional material. The general help will be what the name implies. If they are working with an EWQM and are puzzled as to how to determine the channel slope, they would be able to click on that topic and be given instructions on that parameter. The general help area is one that will be under construction for some time.

Prior to even starting to run SWAT, the student must be able to delineate the watershed. In spite of working previously with contours in an introductory surveying class, this area drew complete blanks from the majority of EWQM students. This personally offended this instructor since I taught the surveying class. I promised myself that these EWQM students would have a firm grasp on this matter. We are fortunate to have at NCA&TSU an Environmental Studies Laboratory that serves as our watershed for SWAT and other EWQMs. To understand the delineation of a watershed, the students are able to; visit the site, examine a 3-dimensional model of the watershed, and compare and work with a 2-dimensional topographic map. The instructional and exercise material on the Internet and for use in the classroom for delineation is:

1. Examining a 3-dimensional model of the Environmental Studies Lab simultaneously with the 2-dimensional topographic map. This would allow them to make general observations of the relationships between the two representations. (http://152.8.7.153/ewqm/swat/instruct/delin)
2. Understanding and interpretation of the basic concepts of contours – basic instruction and exercises

3. Using the same scale 3-dimensional and 2-dimensional models, delineate the obvious ridge lines and boundaries on the 2-dimensional models. It is amazing to see a student's face when a ridge line is evident in a 3-D model.
4. Site visit – using the topographic map, visit the environmental studies lab and delineate the urban portions of the watershed that are not obvious on the topographic map due to man-made structures

General Help

The general help session will be geared to selecting a topic, pointing and clicking to the area where there is a problem and the solution would either popup if short or lead to another location if instruction is needed. This allows two things to happen. Independently, a student could tackle the running of SWAT. We are making no claims as to the accuracy of this approach. Or it may be that the instructional material on determining the channel length was done the week before spring break and the student has completely forgotten it. Therefore, when the student hits that stumbling block, the student could track down detailed instructions on how to determine that parameter. It would be as simple as clicking on the:

- General Help,
- Subbasin icon
- Routing parameter

At that point the instructional material would be available and the problem hopefully solved.

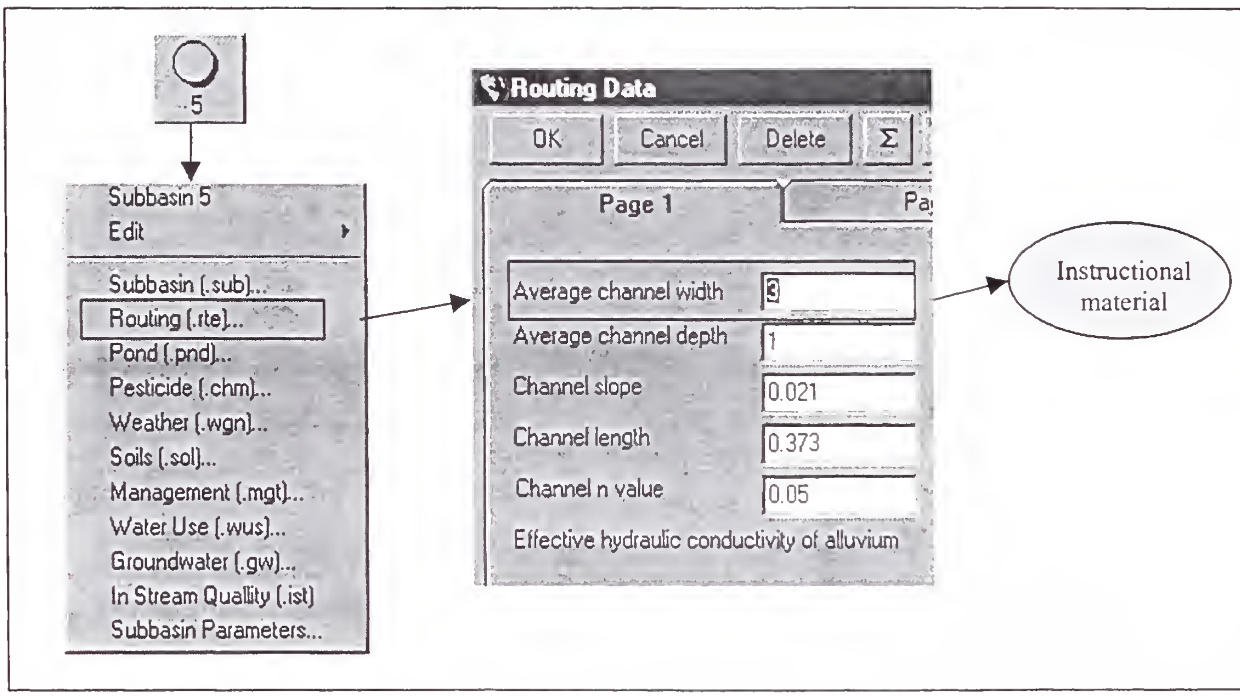

Figure 4 – Guide to help on determining average channel width

Conclusions

In examining the methodologies that we have tried, we have arrived at two areas of agreement. The first is in the teaching approach to EWQMs. Three things need to happen for the undergraduate student to obtain a grasp of a model. First, they need to have a good physical understanding of the watershed area and the parameters that they are modeling. Second, they need to have hands-on computer usage with the model. At this point, instructions for selecting the parameters needs to be available to keep them from getting frustrated. This is accomplished with the EWQM web site. Lastly, they need classroom instruction on the theoretical concepts. With this order of instruction, from broad to refined, the undergraduate student should successfully acquire a basic understanding of EWQMs.

Finally, we found that the involvement of students in a project at the undergraduate level improved retention rates. We feel that the personal involvement of the students in the project, the aspect of constructing physical educational tools as part of the classroom experience and the one-on-one involvement with the faculty have

contributed to this improvement. With success in these areas, we hope to apply these concepts not only in the erosion and water quality modeling course, but in other courses as well.

Acknowledgement

We are grateful to Shouu-Yu Chang, Department of Energy Chair of Excellence in Environmental Sciences Program, College of Engineering, NCA&TSU for providing some financial assistance for this project.

Instructional Approaches and Materials for Teaching WEPP to Undergraduates[1]

Abstract

With the new interface, WEPP can now be incorporated in undergraduate courses. This paper introduces *internet based instructional materials* for teaching WEPP basics (http://152.8.7.153/ewqm/wepp), presents a 'successful' instructional approach for teaching WEPP basics, and discusses why WEPP is an excellent tool for teaching water erosion processes. Instructional materials called 'WEPP Appetizer', contains WEPP Quick Tours, exercises, general help, and physical models. Physical models can be purchased from the ASAE-A&T student chapter.

Keywords

WEPP, instructional materials, water quality modeling, EWQM, soil erosion

[1] Paper No. 992149 – 'An Internet Based Tutorial on the WEPP98 Interface', M. Reyes, Associate Professor, P. Fersner, Adjunct Assistant Professor, and L. Waters, A. Grubbs, E. Phillips, B. Mitchell, and C. Lynch, undergraduates, Agricultural and Biosystems Engineering, North Carolina A&T State University, Greensboro, NC 27411. Email: reyes@ncat.edu and fersner@ncat.edu. Funding was provided by USDA-CSREES Capacity Building Grants Program.

Introduction

The Water Erosion Prediction Project (WEPP) model is a new generation erosion prediction technology for use in soil and water conservation, and environmental planning and assessment. It is based on modern hydrologic and erosion science and is process oriented. WEPP is a continuous simulation model that predicts soil loss and sediment deposition from hillsides and from small channels, and sediment deposition from impoundments (Flanagan and Nearing, 1995).

Compared with the Universal Soil Loss Equation (USLE, Wischmeier and Smith, 1978), WEPP is an extremely complicated model. For example, USLE has six variables, while WEPP has about 1102 (this is the sum of the variables at the back of thirteen chapters in the WEPP technical documentation). Because of its complexity, potential WEPP users are often hesitant to learn the model, hence at present it is primarily used by the research community.

WEPP was formally released in 1995. Because, it had a DOS based interface, WEPP was not user-friendly. However, by 1999 WEPP scientists released in our opinion, a user-friendly interface (WEPP Windows 95/98/NT interface, Beta 3.0 Version, built May 18, 1999). Despite the interface's friendliness, a new user may still find it confusing. Therefore, approaches and materials for WEPP instruction are needed. The objectives of this paper are:

- *To introduce instructional materials for teaching WEPP basics to undergraduates*
- *To present a 'successful' instructional approach for teaching WEPP basics to undergraduates*
- *To discuss why WEPP is an excellent instructional tool for teaching the processes of soil erosion by water to undergraduates.*

We emphasized undergraduates because they will be the new generation of environmental professionals (NGEP) in the next millennium. They will implement the change from old USLE technology to new WEPP technology.

WEPP Instructional Materials

We have been preparing WEPP instructional materials for undergraduates. These are being developed based on our four years experience in teaching erosion and water quality models (EWQM, Fersner, et al., 1999) to undergraduates, and hosting and organizing seven EWQM workshops. The instructional materials are classified as internet based tutorial and physical models.

Internet Based Tutorial

Our internet based tutorial is a modern name for a lesson plan posted in the web (☺). The address is: *http://152.8.7.153/ewqm/wepp*. It is designed to capture the interest of the 'visual' video game generation, a generation propelled by instant gratification. 'WEPP Appetizer' is an excellent title for this tutorial.

Scope of WEPP Appetizer

WEPP Appetizer: will NOT make someone a WEPP expert or a certified 'WEPPEE', is NOT a substitute for WEPP help or documentation, is NOT for seasoned WEPP users or WEPP researchers, is NOT a WEPP FAQ (Frequently Asked Questions) sheet, does NOT explain the details of WEPP science, and does NOT cover all of the WEPP interface.

WEPP Appetizer: will provide instant WEPP gratification for the NGEPs, is a bait for hooking the NGEPs to WEPP, is simple and leaves out the complex maze of WEPP, will be continuously improved, and is composed of a series of short instruction materials and WEPP runs designed for understanding simple WEPP concepts and basic WEPP interface functions. Our hope is some NGEPs will get hooked on WEPP, get WEPP training, and use WEPP as one of their professional tools.

Contents of WEPP Appetizer

WEPP Appetizer is composed of four sections: WEPP quick tours, exercises, general help and physical models (Fersner, et al., 1999).

- *Quick Tours* will concentrate on WEPP technology that we feel undergraduates ought to know, and will understand, like to learn, and enjoy. It contains Quick Tours for a WEPP interface section.
- *Exercises* will parallel the instructional material and provide some 'hands-on' questions that will enhance the learning of a WEPP interface section
- *General help* will provide practical instructions on a WEPP interface section
- *Physical models* contains pictures and descriptions of the models discussed in the next section, ordering procedure, and cost

In Frame 1 is a sample WEPP Quick Tour. WEPP Quick Tours contain a title, objective, expected results after finishing the tour, and three to four links. In Frame 2 are parts of WEPP Quick Tour Number 1 when the links are clicked.

WEPP Quick Tour Number 1
General Overview of the WEPP Interface

Objective: To learn general contents of the WEPP interface

At the end of this session:

- You downloaded WEPP from the web site of the USDA, National Soil Erosion Research Laboratory
- You can identify important parts of the interface

<u>Downloading WEPP Windows 95/98/NT Beta Version 3.0</u>

<u>Identifying important parts of the interface</u>

End of WEPP Quick Tour Number 1

<u>Go to WEPP Exercise 1</u>
<u>Next Tour</u>

Frame 1: Sample WEPP Quick Tour

<u>Downloading WEPP Windows 95/98/NT Beta Version 3.0</u>

1. Click on: **<u>http://topsoil.nserl.purdue.edu/weppmain/weppbeta/weppwin.exe</u>**
2. Choose save this program to disk and click OK.
3. From the Save in: 'window', choose the folder or make a folder of where the *weppwin.exe* file will be located and remember it.
4. From windows explorer go to the folder where *weppwin.exe* is located and double click the file
5. Follow self explanatory installation instructions (when asked, choose the suggested defaults)

<u>Identifying important parts of the interface</u> (Each number is a frame. The frames are not shown to save on space)

1. ⟶ always click on the arrow
2. ⟶ arrow will point on the Windows Start button
3. ⟶ arrow will point on the Programs – WEPP Interface
4. ⟶ arrow will point on the Close button of the WEPP Interface starting frame
5. ⟶ arrow will .... and many more frames

Frame 2: Parts of WEPP Tour Number 1 links

In Frame 3 is a sample WEPP exercise. WEPP exercises contain a title, objective, expected results after finishing the exercise, WEPP navigation, test your skills, and check your answers. Frame 4 shows parts of WEPP Exercise Number 1 when the links are clicked. WEPP navigation has an 'I forgot !!!!' link. 'I forgot!!!!' is linked to the WEPP quick tour topic related to that step.

Physical Models

It is wise not to divorce computer models with physical models. We found that most undergraduates grasp the concepts, variables, and parameters of computer models when physical models are used to explain them. The term 'WEPP hillslope' for example is easier to explain by pointing to the hillside of a 3-D watershed. Therefore, we built physical models of the Environmental Studies Laboratory (ESL).

WEPP Exercise Number 1
General Overview of the WEPP Interface

Objective: To navigate through the major sections of the WEPP interface.

At the end of this session:

- You navigated major parts of the WEPP interface.
- You had a WEPP run.

WEPP Navigation 1

Test your skills 1

Check your answers 1

End of WEPP Exercise Number 1

Return to WEPP Tour Number 1
Go to WEPP Tour Number 2
Next Exercise

Frame 3: Sample WEPP Exercise

WEPP Navigation 1

1. Open the WEPP interface. I forgot!!!
2. Open the project file '*default.prj*'. I forgot!!!
3. Change region info from: '*continuous corn*' to '*my first WEPP run*'. I forgot!!!
4. Change slope file to '*uniform*'. I forgot!!!
5. Change climate file to '*Ithaca_NY*'. I forgot!!!
6. Save the project as *humility.prj*. I forgot!!!
7. ……… and so on. I forgot!!!

Test your skills 1

Print the questions and check your answers later. Open WEPP file default.prj.

1. What is the segment length of the profile (in feet)?
2. Change from Metric to English units. What is the segment length of the profile (in meters)?
3. What is the average annual precipitation?
4. What is the average soil loss?
5. What is the symbol of the WEPPS common look toggle?
6. ………… and so on

Check your answers 1?

Frame 4: Parts of WEPP Exercise Number 1 links

ESL is a small watershed located at the North Carolina A&T State University (A&T) Farm about three miles from the A&T campus. The models are an aerial photograph, topographic map, 3-D model of the watershed, 3-D model of a subwatershed, accessories for the subwatershed model, and 3-D plexi-glass model of a soil profile in the subwatershed. The ASAE-A&T chapter will be selling these physical models and the prices are posted in the ewqm web-site (http//152.8.7.153/ewqm/wepp/forsale). For a negotiated fee, ASAE-A&T can also build a physical model for your site. Just send your specifications, specially the topographic map. You can contact us at our email addresses (reyes@ncat.edu or fersner@ncat.edu) if you are interested in having a physical model built for your site.

WEPP Instructional Approach

Based from our experience in teaching EWQMs to undergraduates (Fersner, et al, 1999), a successful WEPP training requires four instructional approaches. They are: (i) a *visit* to the actual site where WEPP will be used, (ii) an *explanation* of the WEPP parameters using a miniature (physical) model of the site or other physical models, (iii) a WEPP *'hands-on'* computer usage, and (iv) classroom discussions of WEPP science. Begin with a site visit since it gives a mind-set that WEPP is not for the classroom but for the field. After a site visit, do an actual computer usage while explaining WEPP parameters, and encouraging group participation in the discussions of WEPP science. As much as possible, use physical models to explain WEPP parameters.

WEPP an Instructional Tool for Teaching Water Erosion Concepts

WEPP is an excellent tool in explaining water erosion concepts. The four major factors affecting erosion: climate, soil, topography, and land use or land management are distinctly separated in the WEPP interface. Soil loss changes due to changes in these factors can easily be demonstrated by few clicks and choices. The interface gives a visual representation of a hillslope in terms of its: (i) slope length, shape and steepness; (ii) management; and (iii) variations in soil types. Furthermore, soil loss rates within a hillslope are shown by varying shades. The darker the shade color, the higher is the rate. Moreover, average rain, runoff, average soil loss, and sediment loss is shown on a table. Therefore, the instructor has an interesting dynamic visual representation of soil losses on a hillslope which are useful for discussing and explaining water erosion processes. The last WEPP exercise in the web-site illustrates this point.

Summary

With the new interface, WEPP can now be incorporated in undergraduate courses. WEPP undergraduate instructional materials called 'WEPP Appetizers' are available at http://152.8.7.153/ewqm/wepp. It contains WEPP Quick Tours, exercises, general help, and physical models. Visit the web-site and purchase the physical models. A successful WEPP training requires : (i) a *visit* to the actual site where WEPP will be used, (ii) an *explanation* of the WEPP parameters using a miniature (physical) model of the site or other physical models, (iii) a WEPP *'hands-on'* computer usage, and (iv) classroom discussions of WEPP science. Finally, WEPP is a highly recommended tool that you can use for demonstrating water erosion concepts.

Acknowledgement

We are grateful to Shou-Yu Chang, Department of Energy Chair of Excellence in Environmental Sciences Program, College of Engineering, NCA&TSU for providing some financial assistance for this project.

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Wildcat Creek Watershed: An Educational Web Page Utilizing GIS[1]

Abstract

GIS is a computer-based program that can be used to interactively view maps. Using this technology along with the World Wide Web (WWW), these maps can be a useful tool to provide information to people. This paper discusses the methodology used to create a GIS project that showed information about the Wildcat Creek Watershed in North Central Indiana. This watershed was chosen by the Indiana Department of Environmental Management as their pilot project to look at pollution at a watershed scale. This web site will give the ability for stakeholders to learn more information about the watershed.

Using various data sources and ArcView 3.0a, a project was developed for the watershed. The view from the project was served to the Internet using ArcView's Internet Map Server. A web site was developed around this view. Within the web site a 3-D view of the watershed was included, information about the watershed, and other useful links.

Keywords: Geographic Information Systems, World Wide Web, Computer Aided Learning, Wildcat Creek Watershed

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[1] ASAE Paper 99-2150. Alyson Faulkenburg, Graduate Research Assistant, Department of Agricultural and Biological Engineering, Purdue University, West Lafayette, IN 47907 (faulkenb@ecn.purdue.edu).

Introduction

The goal of this project was to create an educational web page of the Wildcat Creek Watershed utilizing GIS data. This web page will serve as an educational tool for people interested in the watershed. The Internet was chosen as the method of delivery due to its increasing accessibility to people. Stakeholders, with access to computers, have the ability to view this information in the privacy of their home or office. Using the Internet as a way to educate people has been found to be an effective way to enhance their knowledge of a subject (Kulic, 1986).

This watershed was chosen by the Indiana Department of Environmental Management (IDEM) as their pilot project to look at water pollution on a watershed scale. As a part of the project, stakeholders within the watershed were asked to become involved in the planning process. This web page could serve as a way for stakeholders to get acquainted with the boundaries of the watershed, learn the types of land uses within, and help them with their decisions regarding the watershed.

The Wildcat Creek Watershed is located in North Central Indiana (Figure 1). The watershed lies in the seven counties of Tippecanoe, Clinton, Carroll, Howard, Tipton, Madison, and Grant. The largest cities within the watershed are Kokomo, Frankfort, and a portion of Lafayette. The watershed covers 803 mi^2 with 93.9% agricultural land, 2.4% residential, 2.6% deciduous forest, and 1.1% other.

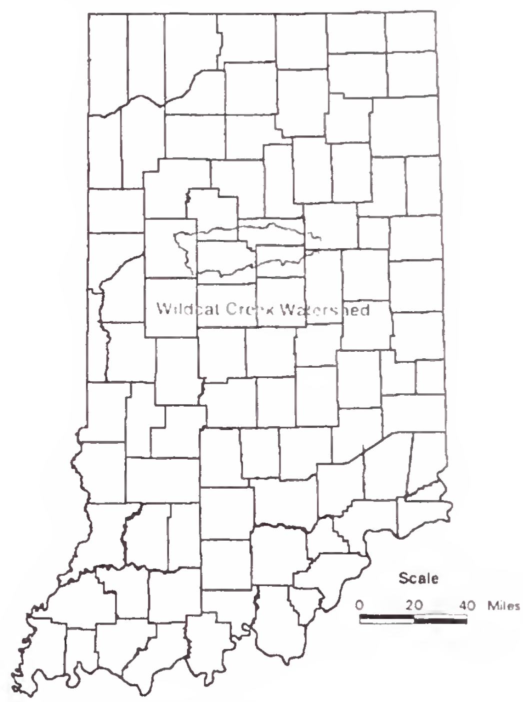

Figure 1: Location of Watershed

A Geographical Information System (GIS) is a computer-based program designed to store, manipulate, analyze, and display sets of data (ESRI, 1995). Unlike paper maps, GIS-based maps allow the user to determine what data they want displayed. The GIS program ArcView 3.0a developed by ESRI was used to develop this project. Along with the standard interface several extensions were utilized.

Methods

Data Utilized

Three main types of data were used. The first type were digital elevation models developed by the U.S. Geological Survey. The maps were grids that had 30m x 30m spacing using Universal Transverse Mercator (UTM) projection.

The second data type used was TIGER® data. TIGER® is an acronym for Topologically Integrated Geographic Encoding and Referencing. This system and digital database was developed at the census bureau to support their needs for the census and other programs. The type of TIGER® data used was the road themes for the seven counties. These data sets were used because the attributes tables held the names of the roads. This data was developed at a scale of 1:100000.

The final data type used was obtained from the Indiana Natural Resources and Environmental GIS data CD-ROM compiled by Dr. Engel and associates (Engel et al., 1997). The themes used were HUA's (hydrologic unit areas), rivers, county outlines, cities, land use, highways, and railroads. These data sets were developed at a scale of 1:250000. The hydrologic unit areas and the land use themes were developed by the USGS. The highways, railroads, and rivers were obtained from the USGS's digital line graphs. The cities theme was obtained from the Department of Transportation.

Interface Changes

There were four main changes to the ArcView interface. The first change was the extension Projector!. This extension allows the changing of a shapefile's projections without going into ARC/INFO. The shapefiles from the Indiana Environmental C.D. were reprojected from a projection of UTM, datum of NAD27, and spheroid of Clarke1866 to the projection of UTM, datum of NAD83, and spheroid of GRS80.

The second change made was the adding of the Xtools extension. This extension creates several commands that are equivalent to ARC/INFO commands. Some example commands are buffer, clip, intersect, merge, union, and update. The clip command was utilized several times.

The third type of change was 3-D Analyst. This extension was developed to allow the ability to create, visualize, and analyze 3-D surfaces. The DEM grid was converted into 3-D with the rivers overlaid.

The final interface change was the Internet Map Server (IMS). This extension was created so that GIS data can be interactively served to the public. This was used to allow people access to the data in the project. The extension uses user friendly pop up windows to step the user through the set up. Once the user chooses the components needed a web page or portion of a page is created. For this project the "snippet" was chosen and copied into a page that had already been created. A snippet is the term used to describe the Internet Map Server view.

Procedure

The first step in the analysis was to combine the DEM's. At the 1:24000 scale, 36 grids were needed for the watershed area. The grids were combined into one using the merge command in ARC/INFO. Places that contained no data were filled using the fill command.

The next step was to reproject the shapefile data from NAD27 to the DEM and TIGER® data projection of NAD83. Here the projector! Extension previously discussed was used.

Once all the themes had the same projection they were clipped to the size needed. The DEM was clipped down to the watershed boundary using the map calculator. The data obtained from the Indiana Environmental C.D. was available in two versions, county by county or statewide. Using the clip command, in the Xtools extension, the statewide data was clipped to the needed boundaries. First the 8, 11, and 14 digit watersheds were selected and converted into a shapefile. Using these boundaries the land use and rivers shapefiles were then clipped. The seven counties were then selected and also converted into a shapefile. The themes of highways, railroads, and cities were then clipped down to the counties boundaries.

Next the attribute tables were edited. This was done so that the identify tool could be used on the web page. The land use table was edited to show the land use type names. The cities and counties were also edited to show their respective names. The TIGER® roads shapefile attribute table already included the road names the only change needed was to change the properties to show only the names.

Once the tables were completed, the next step was to portray the watershed in 3-D. The themes were copied into 3-D Analyst and manipulated (Figure 2). A Z factor of 10 was used to give the grid depth. A larger factor could have been used for Indiana however it would give the appearance of mountains instead of rolling hills. The rivers were set to a Z factor of 10.5 with an extruding factor of 2.0; this will enable the viewer to see the steams more clearly.

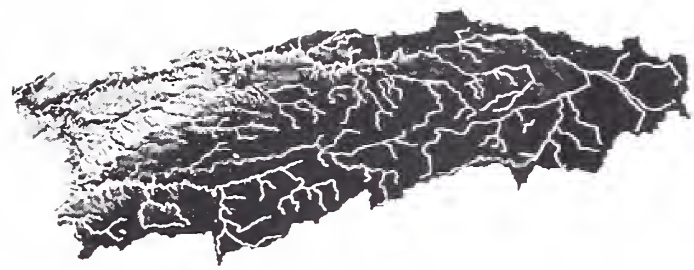

Figure 2: 3-D View of Watershed

The 3-D views of the watershed could not be displayed using the Internet Map Server. To display the view on the WWW an animated GIF file was created. To create this file a 3-D view was saved as a GIF file and then the watershed was moved and another GIF file was saved. This was repeated 34 times. These GIF files were then combined into one animated GIF file with each view lasting 0.5 seconds. This was done using Adobe® ImageReady™ 1.0.

The final step was to create the Internet Map Server snippet that was previously discussed in the interface changes. This snippet can be seen in Figure 3.

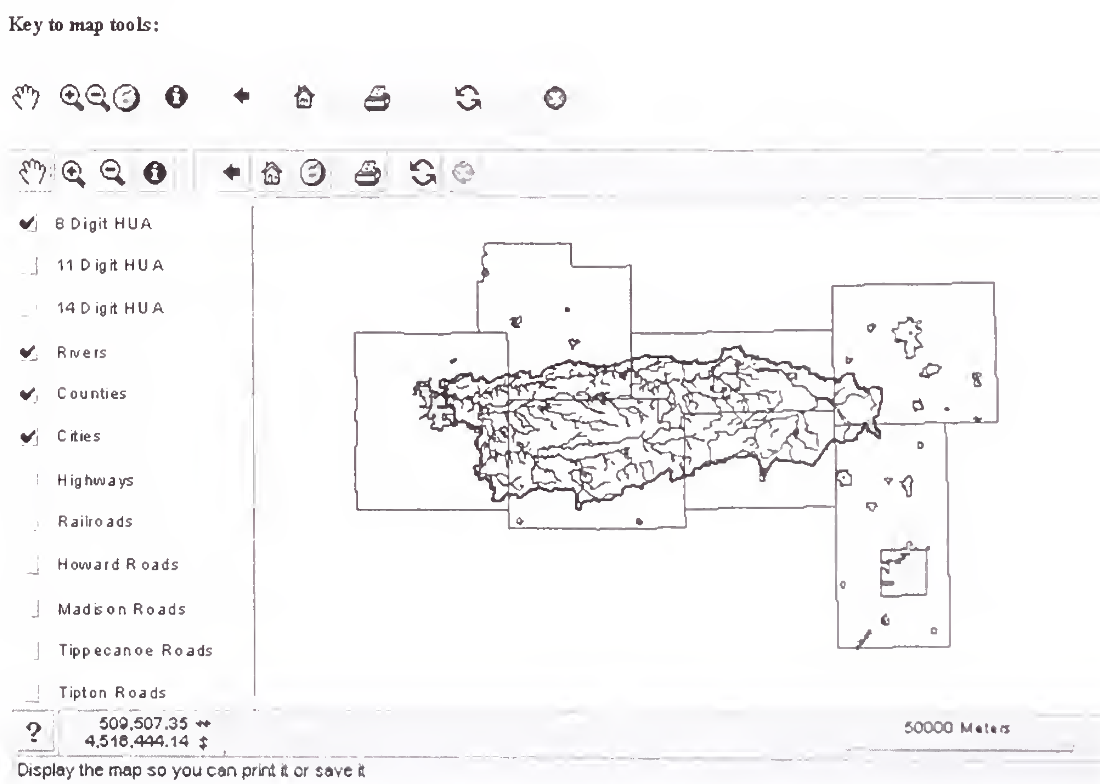

Figure 3: Internet Map Server Snippet

Results

The web site (http://abent0.ecn.purdue.edu/gis590/Wildcat/main.html) consists of five pages.

- The first page is a description of what a watershed is. This page was created because the target audience of this project was people who may not realize that they live in a watershed or even what a watershed is. This definition also included an explanation on why watersheds are important.
- The second part of the web site was a description of the Wildcat Creek Watershed. Included was the size of land, miles of streams, and counties that make up the area. Also included were the facts that over 20 of the stream segments do not support life and than many TMDL's are exceeded.
- The next part of the web site was the "fly over" of the watershed. This consisted of the animated GIF of the 3-D views. When looked at on the web page it gives the appearance of flying over the watershed from the headwaters to the mouth of the rivers.
- The next page was the Internet Map Server page. Using the snippet created by the Internet Map Server extension a web page was developed. This snippet allows the user to interactively select themes and to obtain information about the themes. An example of the available information is the TIGER® roads. The user can select the information tool and then select a road; a new page will open giving the name of the road. The user can also turn on or off any of the available themes and zoom in or out. A key to the various icon buttons was also made and is shown above the map area.
- The final portion of the site is a page with links to other information sources. This page lists several pages that can be used to obtain more information about watersheds. Some of the links include the EPA, IDEM, and the electronic watershed page for Indiana.

The various watershed protection groups within the Wildcat Creek Watershed can use this web page. They can use it to help their efforts in recruiting volunteers and to educate the people within the watershed. It can also be used by governmental officials in their decision making. A final possibility for the web page could be other watershed groups. They can use this page as a model on how to get their information out to the public.

Conclusion

This project integrated the technologies of the WWW and GIS to educate stakeholders of a watershed. With the available data a GIS project was developed that could then be served to the Internet. Along with this GIS data, a 3-D view of the watershed and other information was added to the web site. This information can then be used by various government agencies and educational institutions to show people what their watershed looks like and why they should be interested in it.

This web site is a good starting point on using the Internet as an educational tool. With rapidly advancing technology this project can be advanced to include more information about the watershed. Included could be volunteer's monitoring data, USGS streamgaging data, and updates on watershed meetings. Overall there is a great potential for the use of this type of information.

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TIGER data are available at http://www.census.gov/geo/www/tiger/

USGS data are available from the USGS web site: http://edcwww.cr.usgs.gov/webglis

Evolution and Evaluation of Nutrient Enabled NAPRA WWW Decision Support System[1]

Abstract

The nutrient enabled National Agricultural Pesticide Risk Analysis (NAPRA) WWW Decision Support System (http://danpatch.ecn.purdue.edu/~napra/) was enhanced to provide an improved user-friendly environment by adding more online help using JavaScript, Java and data in the ORACLE relational database. For example, the weather station map of Indiana is provided so that the user can select the appropriate weather data for the region they would like to evaluate. Pesticide properties data from USDA was incorporated in an ORACLE database so users can choose the pesticide of interest by selecting either trade name or common name. Based on the pesticide name the user selects, NAPRA WWW queries obtain the pesticide attributes from the ORACLE database for the NAPRA pesticide component input. Toxicity level information for human and aquatic creatures is also stored in the ORACLE database. For pesticide and nitrogen losses, 50% and 10% probability of exceedence values for surface water, shallow groundwater and sediment losses are provided in a table.

Since the NAPRA WWW system showed promise for identifying critical watersheds from a nutrient loss perspective, the accuracy of the NAPRA predicted nitrogen loss results was evaluated further by comparing the results with observed USGS well nitrate concentration data within Indiana. The results indicate the NAPRA WWW system is able to identify areas where elevated nitrogen levels occur in wells.

Keywords: GLEAMS, NAPRA, Nutrients, Vulnerability, Water Quality, Decision Support System, Models.

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Introduction

Groundwater is an important resource in much of the midwestern USA. More than 95% of the people in rural areas and more than 50% of the people in urban areas use groundwater as a primary source of drinking water. Thus, protecting ground water in the midwestern USA is of great significance (US EPA, 1992). Excessive levels of nitrate or nitrite - concentrations of nitrate which exceed 10 milligrams per liter (ppm), or nitrite which exceeds 1 milligram per liter - in drinking water can be lethal to infants under six months of age (US EPA, 1992). Thus, it is essential to minimize NPS pollution in areas that groundwater is vulnerable. Regional vulnerability to NPS contamination maps for efficient groundwater management and protection are also needed.

As the nature of agricultural NPS pollution is essentially spatial, it seems appropriate to use a spatial analysis system to analyze and model the data related to the problem. However, even models integrated with GIS still require experts to operate them in most instances. The nutrient enabled NAPRA WWW system was created for such use, but was somewhat hard for potential users to operate (Engel, 1998).

The objectives of this project were:

1) to enhance the nutrient enabled NAPRA WWW Decision Support System interface by incorporating more database support and GIS maps.
2) to evaluate the nutrient enabled NAPRA WWW system further by comparing the results with observed USGS well nitrate concentration data within Indiana.

Review of Literature

To estimate groundwater vulnerability to agricultural NPS pollution in Indiana, DRASTIC, SEEPAGE, and SPISP have been used (Engel et al., 1996). DRASTIC considers various hydrogeological settings which influence the pollution potential of a region. DRASTIC is based on seven major factors; 1) Depth to water, 2) net Recharge, 3) Aquifer media, 4) Soil media, 5) Topography, 6) Impact of the vadose zone and 7) hydraulic Conductivity of the aquifer. The System for Early Evaluation of Pollution potential of Agricultural Groundwater Environments (SEEPAGE) considers various hydrogeological settings and physical properties of the soil that affect the groundwater vulnerability to pollution potential. Soil Pesticide Interaction Screening Procedure (SPISP) considers pesticide properties and soil properties in predicting vulnerability of water. The DRASTIC, SEEPAGE, and SPISP approaches for predicting groundwater vulnerability performed reasonably well for both nutrients and pesticides. The maps resulting from these indices are being used to target groundwater protection efforts within Indiana (Engel, 1996).

The WWW-based NAPRA approach (Engel et al., 1998) was developed to estimate the site specific effects of land use and management on water quality with respect to pesticides and thus complements DRASTIC, SEEPAGE and SPISP. This NAPRA approach is designed to be easy to use and widely accessible through the WWW (Engel et al., 1998). The nutrient component of GLEAMS was added to the NAPRA WWW system to develop a nutrient loss decision support system to identify critical watersheds in Indiana and the Midwest from a nutrient management perspective. Nutrient enabled NAPRA WWW showed potential for identifying critical watersheds from a nutrient loss perspective (Lim, 1998).

Methodology

The nutrient enabled NAPRA WWW interface was modified using JavaScript, Java, and Dynamic HTML to create a more user-friendly interface. The most significant enhancements are described herein. SSURGO soil data for 6 counties in Indiana were incorporated into NAPRA WWW to facilitate use at a local level.

The JShape applet (JShape, http://shiuhlin. ml.org) allows ArcView shape files, grids and other images to be displayed in WWW browsers. All unique combinations of the Theissen weather station polygons and counties in Indiana were created using "r.cross" command in the GRASS GIS. The JShape applet is used to serve the county map with Thiessen weather station polygons. An external module added to JShape allows the user to select the weather station directly from a map by selecting their location in a county, so weather station information can be transferred to the NAPRA WWW main interface directly and also checks for errors.

The pesticide properties database from USDA was incorporated in the ORACLE database so the user can choose the pesticide of interest by selecting either trade name or common name. Based on the pesticide name the user selects, NAPRA WWW obtains the pesticide attributes such as soil half-life, partitioning coefficient (koc), water solubility, foliar residue half-life, and fraction of pesticide on the foliage available for wash-off by rainfall from the

ORACLE database for the NAPRA pesticide component input. Depending on nutrient types selected, the units of application and application methods are updated on the fly so the user selects a legal option from the menu.

The previous NAPRA WWW system didn't allow multiple crop rotation simulations, although crop rotations are common in reality (Lim, 1998). Thus, the simulation results from one continuous crop and a second continuous crop had to be combined according to their relative percentage to approximate crop rotation. Crop rotation simulation has been enabled in the new NAPRA WWW system with a maximum of 5 crops in the rotation, but it can be easily extended to a larger number of crops.

The toxicity information for human and aquatic creatures was obtained from USDA NAPRA PPD and stored in an ORACLE database. Six pesticide toxicity levels - 25%, 50%, 100%, 200%, 500%, and 1000% of the standard toxicity level - can be plotted in the pesticide loss output graphs. There are four options – No toxicity, Human toxicity level, Fish toxicity, and Human & Fish toxicity. The toxicity level can guide the user in seeing how much the estimated pesticide concentration exceeds the standard toxic level for human and aquatic creatures.

Once the NAPRA WWW system is run, 50% and 10% probability of exceedence values for surface water, shallow groundwater and sediment losses for pesticide and nitrogen losses are provided in a table to provide a quick understanding of the model results. The updated output interface (developed using JavaScript) is divided into three parts; 1) Model input/output file, 2) Pesticide output in graphical representation, and 3) Nutrient output in graphical representation.

NAPRA WWW predicted nitrogen losses to shallow groundwater are shown in Figure 1. The NAPRA WWW nitrogen leached to shallow ground water values were classified into 4 groups using the Natural Break classification method: Low, Moderate, High and Very High. US Geologic Survey well nitrate concentration data were also divided into 4 groups: Low 0-2 ppm; Moderate 2-5 ppm; High 5-10 ppm; and Very High >10 ppm.

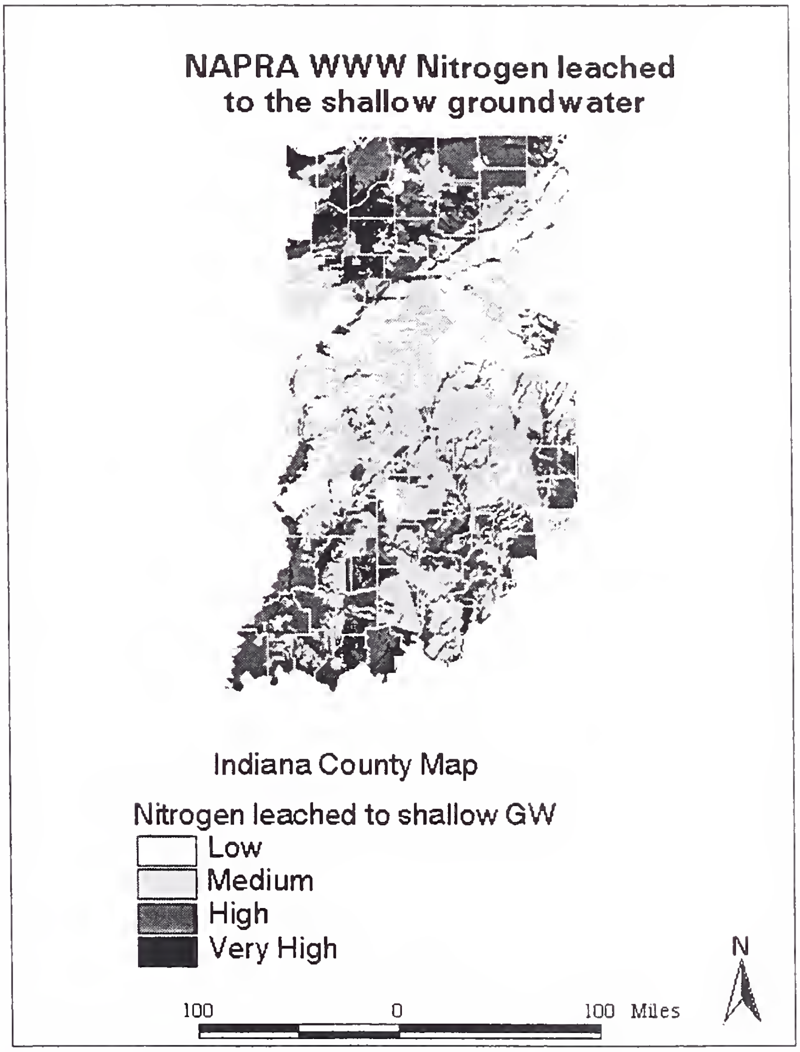

Figure 1. NAPRA WWW predicted nitrogen loss to shallow groundwater

Results

The National Agricultural Pesticide Risk Analysis (NAPRA) WWW system was modified and updated (http://danpatch.ecn.purdue.edu/~napra). A JShape module was added to let the user select the weather station directly from a map. The user can select the weather station by selecting a region in the county map. JShape also allows the user to zoom in, zoom out and query the map. Figure 2 shows the JShape module in the nutrient enabled NAPRA WWW system.

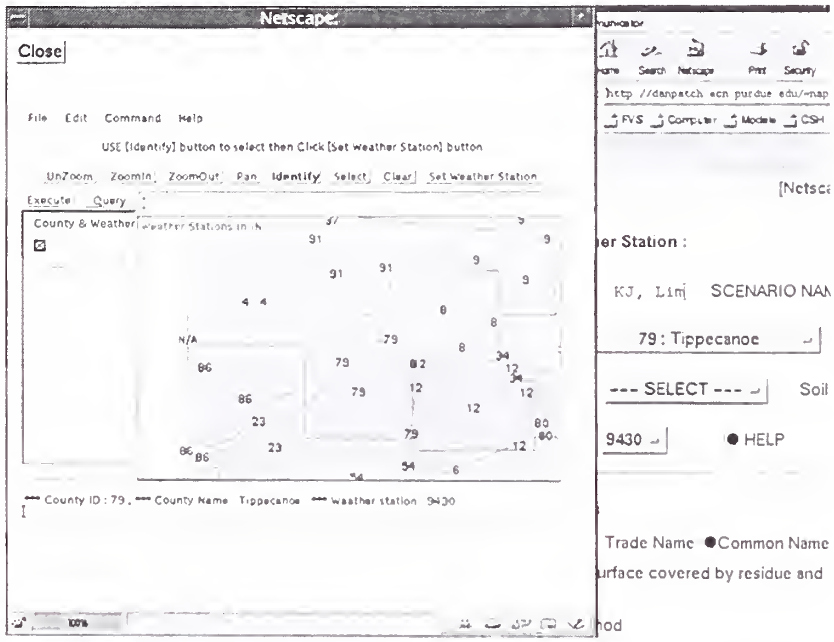

Figure 2. Weather station map with county boundaries using JShape

Figure 3 shows the method for querying pesticide properties stored in the ORACLE database. Pesticide properties of the trade name are queried with the corresponding pesticide common name.

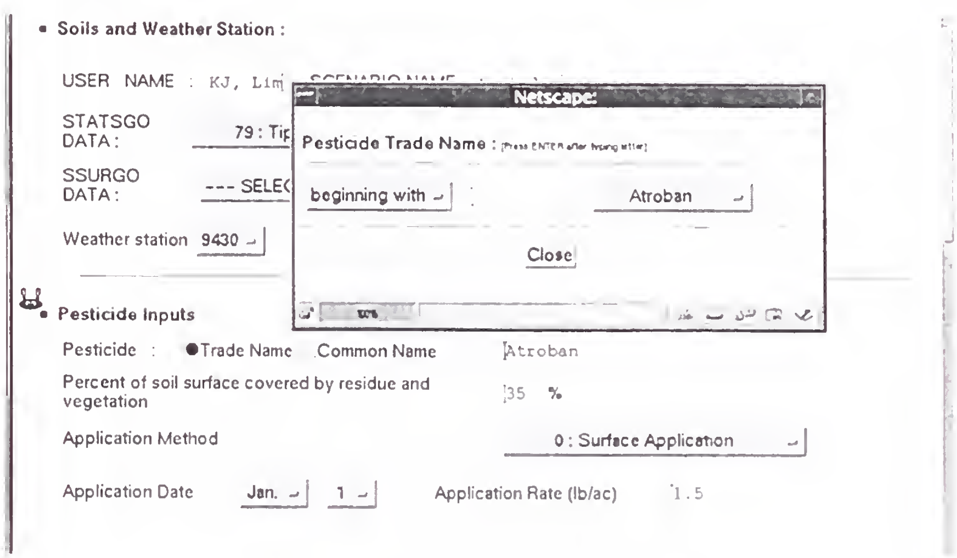

Figure 3. Pesticide trade name query

Online help provides brief descriptions about each step required to run the NAPRA WWW system. Figure 4 shows the NAPRA WWW output interface. The user can view and download GLEAMS Input/Output files for further analysis. Pesticide and nutrient output graphs are provided in JPEG format so the user can save and use them for reporting purposes and further analysis.

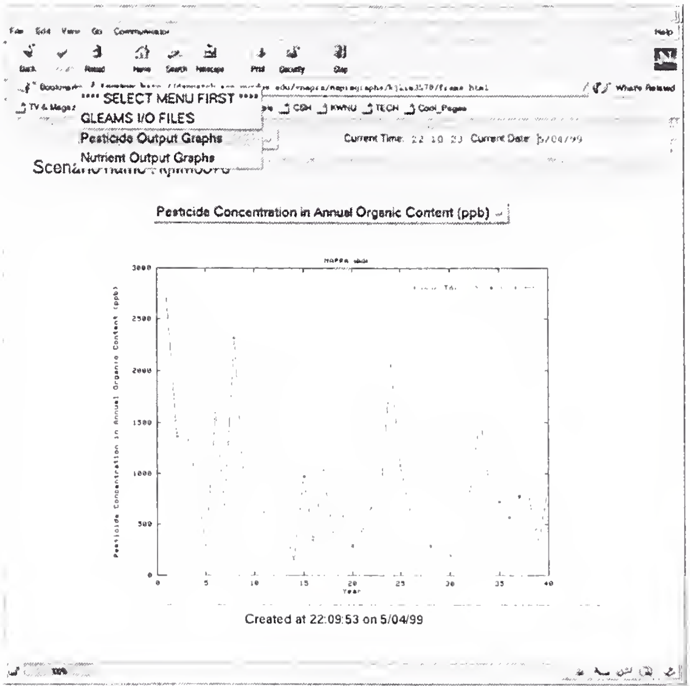

Figure 4. The NAPRA WWW output interface

Figure 5 shows the NAPRA WWW output probability reports. 50% and 10% of pesticide and nutrient probability of exceedence values for surface, shallow groundwater and sediment losses are reported in the table.

50% and 10% Pesticide and Nutrient Probability of Exceedence Values for Surface, Shallow Groundwater and Sediment Losses

NAPRA Predicted Mass Lost	Runoff		Shallow Groundwater		Sediment	
	50 %	10 %	50 %	10 %	50 %	10 %
Atrazine (g/ha)	52.9983	133.9521	2.2054	6.8346	0.2552	0.5036
Nitrogen (kg/ha)	5.2383	8.0376	0.1197	0.2750	0.0054	1.7130
Phosphorus (kg/ha)	0.0000	0.0000	0.0029	0.0048	0.2197	25.6991
NAPRA Predicted Concentration Lost	**Runoff**		**Shallow Groundwater**		**Sediment**	
	50 %	10 %	50 %	10 %	50 %	10 %
Atrazine (ppb)	18.5084	42.8794	1.3734	3.7310	---	---
Nitrogen (ppm)	2.0001	2.1271	0.0867	0.1726	---	---
Phosphorus (ppm)	0.0000	0.0000	0.0017	0.0025	---	---

Figure 5. NAPRA probability result

The NAPRA WWW estimated nitrogen leached to shallow ground water data was compared with the US Geologic Survey well nitrate concentration data. The NAPRA predicted results for each USGS nitrate concentration data location were obtained using a spatial join in a GIS. The comparison of NAPRA WWW nitrogen leached results with observed nitrate concentrations is shown in Table 1.

Table 1. Comprarison of NAPRA WWW nitrogen leached to shallow groundwater with observed nitrate concentrations in wells.

Observed	The NAPRA WWW predicted results			
	Low	Moderate	High	Very High
Low	93	98	139	50
Moderate	10	27	54	15
High	10	20	40	5
Very High	6	2	128	34

Based on the results shown in Table 1, NAPRA WWW has the potential to identify areas within Indiana that are vulnerable to NPS nitrogen reaching groundwater. NPARA WWW correctly categorized 84% of the High and Very High observations even though NAPRA WWW estimated High and Very High potential for nitrogen loss to groundwater in only 45 % of the state.

The NAPRA WWW estimates of nitrogen losses are generally in more severe categories compared to observed nitrate levels. This is expected and can be explained as follows:

1) The NAPRA WWW predicted results indicate there is a *potential* of NPS loss in that region.
2) The NAPRA WWW predicts nitrogen leached to the shallow groundwater, but this doesn't guarantee it will reach wells.

There were few (16%) observations of High and Very High concentrations when NAPRA predicted Low or Moderate potential for nitrogen reaching shallow groundwater. The USGS observed nitrate concentration data was collected in regions with concern about groundwater quality and thus likely contain higher concentration of nitrogen than would random samples. Point sources of nitrogen and factors other than those considered in NAPRA WWW may be responsible for some of the improperly predicted nitrogen detections too.

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Summary and Conclusions

The enhanced NAPRA WWW system provides an improved user-friendly environment. This was accomplished using JavaScript, Java and the ORACLE relational database. Weather station maps are served using JShape to allow the user to select the location of weather stations from maps. The pesticide properties database from USDA and toxicity level information for human and aquatic creatures were incorporated in the NAPRA WWW system. Depending on the pesticide name the user selects, NAPRA WWW obtains the pesticide attributes from the ORACLE database for the NAPRA pesticide component input. Pesticide toxicity levels can be plotted in the pesticide loss output graphs. For pesticide and nitrogen losses, 50% and 10% probability of exceedence values for surface water, shallow groundwater and sediment losses are provided in a table.

The accuracy of the NAPRA predicted nitrogen loss results was evaluated by comparing the results with observed USGS well nitrate concentration data within Indiana. The NAPRA WWW predicted results matched observed nitrate concentrations reasonably well if limitations of NAPRA WWW system are considered. This system has the potential to identify areas where groundwater is vulnerable to nitrogen NPS pollution.

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Evaluation of VFSmod, a Vegetative Filter Strips Hydrology and Sediment Filtration Model[1]

Abstract

VFSmod is a field scale, mechanistic, storm-based model designed to simulate the routing of the incoming hydrograph and sedimentograph from an adjacent field, through a vegetative filter strip (VFS) and to calculate the VFS's outflow, infiltration and sediment trapping efficiency.

VFSmod is a numerical 1-D model that handles time dependent hyetographs, space distributed filter parameters (vegetation roughness or density, slope, infiltration characteristics) and different particle size of the incoming sediment. Any combination of unsteady storm and incoming hydrograph types can be used.

The model has been field tested for different soil and climatic conditions in the uplands of the Piedmont and Coastal Plain areas of North Carolina and can be used as a design tool for VFS.

Keywords: Vegetative filter strips, buffer strips, grass strips, model, overland flow, finite elements, sediment.

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Objective and Scope of the Model

VFSmod (Vegetative Filter Strips model) is a field scale, mechanistic, storm-based model that describes the overland water flow and sediment filtration (transport and deposition) along a vegetative filter strip (VFS) adjacent to a field, used as an off-site non-point source pollution control practice. The model handles time dependent hyetographs, space distributed filter parameters (vegetation roughness or density, slope, infiltration characteristics) and different particle size of the incoming sediment. Any combination of unsteady storm and incoming hydrograph types can be used.

The *VFSmod* model is a software package developed by the Biological & Agricultural Engineering Department at N.C. State University and the Instituto Canario de Investigaciones Agrarias (ICIA). The current version of the model integrates a numerical overland flow submodel (Muñoz-Carpena et al., 1993a,b) coupled with a sediment filtration model through grass strips developed at the University of Kentucky (Wilson et al, 1981). A new module to handle sediment sorbed contaminants, such as phosphorous, and a Windows interface are under development and will be added in future versions of the model.

The program is written in standard FORTRAN 77 and can be run either under UNIX or MS-DOS platforms.

Components, Processes and Solution Techniques

VFSmod consists of a series of modules simulating the behavior of water and sediment in the surface of the VFS. The current modules available are (Figure 1):

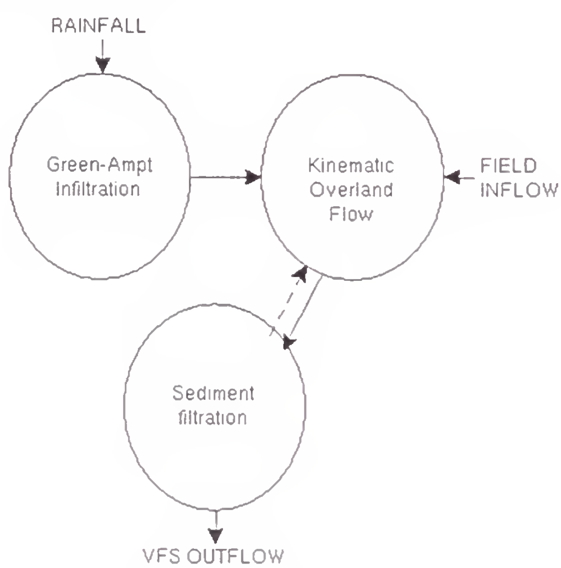

Figure 1. Components of the model

i) Green-Ampt infiltration module: a module for calculating the water balance in the soil surface; ii) kinematic wave overland flow module: a 1-D module for calculating flow depth and rates on the infiltrating soil surface; iii) sediment filtration module: a module for simulating transport and deposition of the incoming sediment along the VFS.

VFSmod is essentially a 1-D model for the description of water transport and sediment deposition along the VFS. The model can also be used to describe transport at the field scale (or field edge) if flow and transport is mainly in the form of sheet flow (Hortonian) and the 1-D path represents average conditions (field effective values) across the VFS.

The *VFSmod* model uses a variable time step, chosen to limit mass balance errors induced by solving the overland water flow equation. The time step for the simulation is selected by the kinematic wave model to satisfy convergence and computational criteria of the finite element method based on model inputs (Muñoz-Carpena et al., 1993a,b).

The model inputs are specified on a storm basis. State variables are integrated after each event to yield storm outputs.

The Green-Ampt Infiltration Module

The water balance at the surface, rainfall excess ($i_e = r(t) - f(t)$ in Figure 2), is calculated from the storm hyetograph for each node at the domain and every time step.

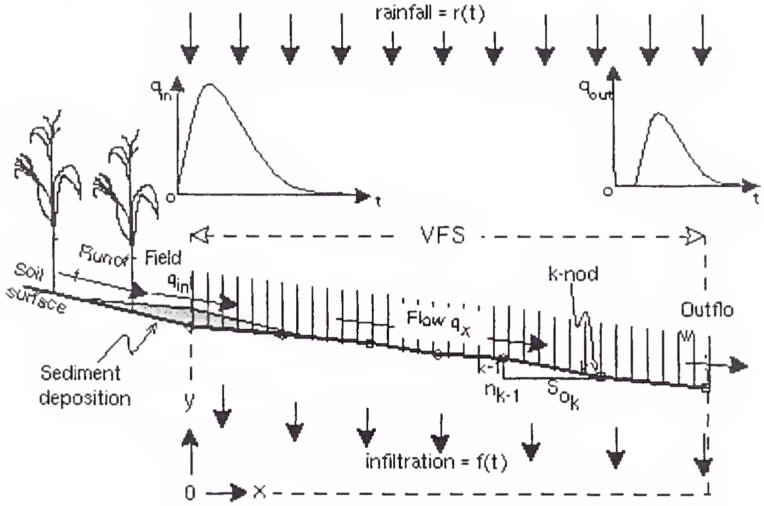

Figure 2. Domain discretization (Muñoz-Carpena et al., 1999)

The calculation is based on a modification of the Green-Ampt equation for unsteady rainfall that considers unsteady rainfall and water balance at the surface (Chu, 1978; Mein and Larson, 1973; Skaggs and Khaheel, 1982; Muñoz-Carpena et al., 1993b).

The Kinematic Wave Overland Flow Module

The program solves the overland flow kinematic wave approximation (KW) of the Saint-Vennant's equations for the 1-D case (Lighthill and Whitham, 1955) in the domain represented in Figure 2.

The incoming hydrograph representing runoff from the adjacent field is input as a time dependent boundary condition at the first node of the finite element grid. The program allows for spatial variation of the parameters n (Manning's surface roughness) and S_o (slope) over the nodes of the system (Figure 2). This feature of the program ensures a good representation of the field conditions for different rainfall events. The model can be operated to provide information on the effect of soil type (infiltration), slope, surface roughness, filter length, storm pattern and field inflow on VFS performance (i.e. reduction of the runoff peak, volume and velocity) (Muñoz-Carpena et al., 1993b). It also describes the flow rate (q), velocity, and depth components throughout the filter for each time step.

The classical numerical solution for KW is subject to kinematic shocks, or oscillations in the solution that develop when a sudden change in conditions (slope, roughness or inflow) occurs. When linking the kinematic wave and the sediment transport models, the soil surface conditions are also changed for each time step, further increasing the potential for the kinematic shock problem. *VFSmod* implements a Petrov-Galerkin formulation (non-standard) finite element to solve the KW equations. This solution procedure reduces the amplitude and frequency of oscillations with respect to the standard Bubnov-Galerkin method (Muñoz-Carpena et al., 1993a), thus improving the model stability and the sediment transport predictions which depend on overland flow values.

The Sediment Filtration Module

The hydrology model is linked to a model for filtration of suspended solids by artificial grass media. This submodel is a based on the theory of the model developed and later tested for field conditions at the University of Kentucky (Barfield et al. 1979; Hayes et al., 1979, 1984; Tollner et al., 1976, 1977, Wilson et al, 1981). It is based on the hydraulics of flow, transport and deposition profiles of sediment in laboratory conditions. The model presents the advantage of being developed specifically for the filtration of suspended solids by grass.

The original sediment model uses a simple approach to calculate flow conditions at specific points of the filter and does not consider the complex effects of rainfall, infiltration, and flow delay caused by the buffer strip.

VFSmod provides a more accurate description of the flow conditions from the hydrology submodel whereas changes in surface conditions (topography, roughness) due to sediment deposition during the event are obtained from the sediment filtration submodel.

Model Inputs and Outputs

The program reads inputs (model parameters and model input variables) from external ASCII-files, which can be prepared from given examples using a conventional text editor. A summary of the model inputs is given in Table 1.

Table 1: Inputs for the model

Class	Inputs
Green-Ampt infiltration	Rainfall hyetograph, soil saturated hydraulic conductivity, soil saturated water content, soil initial water content, soil suction at the wetting front, and surface storage.
Overland flow	Field inflow hydrograph, filter length, filter width, nodal slopes and Manning's roughness across the filter.
Sediment filtration	Modified grass Manning's roughness, Manning's roughness for bare soil, incoming sediment characteristics (median particle size, weight density, fall velocity), effective filter media spacing and height, porosity of deposited sediment, incoming sediment inflow concentration (sedimentograph), and proportion of fine sediment.
General	Number of nodes for the domain, Courant number for numerical solution, total time for the simulation.

The *VFSmod* manual gives literature values for some of these parameters when no field measurements are available. In the case of the soil hydraulic and sediment parameters, these can be chosen from soil texture using tables from the manual.

As an aid to set up the model inputs, the distribution package includes an utility, UH, that creates synthetic model inputs based on the SCS design storm for a given location and soil type. The utility implements the SCS curve number, unit hydrograph and MUSLE concepts to produce ready-to-use input files for *VFSmod*. These inputs are rainfall hyetograph, field inflow hydrograph and field sediment inflow and characteristics.

The program writes output into ASCII-files. Each aspect of the model is written to different files. The model outputs include: information on the water balance (volume of rainfall, field inflow, filter outflow and infiltration), hydrograph, sediment balance (field inflow, filter outflow and deposition), sedimentograph, filter trapping efficiency, and sediment deposition pattern within the filter. The output files contain summaries of the main state variables in the program.

The name of the input file set to process is selected at the command line, and the output file set is created automatically using the name given as input. In this way, different problems can be run from the same directory without overwriting previous results.

Sensitivity Analysis of the Model.

A sensitivity analysis was performed to gain some insight in the dependence of model outputs on certain model parameters and to assist in the model calibration (Muñoz-Carpena et al. 1999). The study showed that the main parameters controlling the hydrology outputs were soil hydraulic conductivity and initial water content whereas the model was fairly insensitive to changes in saturated water content and suction at the wetting front values. Previous research (Muñoz-Carpena et al., 1993a) showed that Manning's surface roughness controls mainly the time to peak of the outgoing hydrograph.

Testing on the sediment component of the model showed that the main parameters controlling sediment outflow are media spacing, and particle diameter. Variations in the modified Manning's roughness had relatively little effect on the output and the effect of media height was only visible for large events when the filter began to be inundated with sediment.

Previous Testing and Applications

The model was tested with natural events data at a North Carolina Piedmont (Muñoz-Carpena et al, 1999) and a Coastal Plain (Muñoz-Carpena, 1993) experimental sites. Both sites had grass filter strips (mixture of fescue, bluegrass and bermuda grass) with ratios of field to filter lengths from 4.5:1 to 9:1. The field area had varying slope from 5-10% and the filter strip somewhat less. The soil types were Cecil clayey at the Piedmont site and Rains loamy-sand at the Coastal Plain site (Parsons et al., 1991). In general, good agreement was obtained between observed and predicted hydrology and sediment outflow values. Some sources of variability were discussed to explain some anomalous events.

A design procedure for VFS using the model along with the utility UH is currently under development (Suwandono et al., 1999). This procedure uses the utility UH to generate runoff hydrographs and sediment losses from upslope source areas to enable testing and evaluating the effectiveness of various VFS scenarios.

Potential Users and Applications of the Model

Since *VFSmod* is a research model, potential users are modelers and scientists involved in studies of sediment pollution from agricultural sources and its control, with the aim to gain a better understanding of the processes involved for a given scenario.

Results from this model can be used, after calibration and field testing in extrapolation or prediction studies for decision making and design (Suwandono et al., 1999). Integrated design procedures may enable extending the model users to include others involved in the design and implementation of VFS.

Known Limitations and Applicability of the Model

Known Limitations of the Model

- The handling of overland flow as sheet flow could pose problems when a filter is not properly maintained and concentrated flow occurs within the filter.
- Since parameters to describe hydrology and sediment transport in VFS are highly variable, field variability is an inherent source of error. A range of variation in the saturated conductivity parameters is usually needed to fit the model to observed data. Although this variation can be explained by changes in surface conditions due to seasonal and biological factors, these changes are difficult to quantify in field situations.

Future Research

- A revision of the sediment filtration submodel to handle particle size distribution explicitly.
- A submodel to handle sediment sorbed contaminants (such as phosphorous).
- An improvement to model usability by the addition on a graphical user interface (GUI).
- Improvement of parameter estimation techniques, calibration procedure and reduction of the uncertainty ranges of sensitive parameters.

Distribution and Training

The model is provided free of charge to qualified users as an educational and research tool. The model and documentation can be downloaded from the internet (in the USA http://www3.bae.ncsu.edu/*VFSmod*/ and in Europe http://www.icia.rcanaria.es/*VFSmod*/) or obtained from the authors. Limited support is available from the authors.

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A Foundation for Modeling Bacterial Contamination of Surface Waters Due to Nonpoint Sources[1]

Abstract

The United States Environmental Protection Agency estimated in 1998 that 19,811 river miles are impaired by pathogens, which is approximately 8% of the nation's impaired streams. Pathogens are disease-causing organisms that may be viral or bacterial in nature. Regulations for human ingestion or contact with water, however, are rarely based directly on pathogen concentrations; rather, indicator organisms, such as fecal coliform bacteria, are used to set standards due to the relative ease with which samples can be collected and enumerated. The objectives of this paper are 1) to compile information related to processes important for modeling bacteria from nonpoint sources (NPSs) such as bacterial growth/die-off, partitioning of bacteria between water and soil, and the effects of best management practices, 2) to evaluate the capabilities of existing bacterial models to represent these processes, and 3) to identify areas for future research.

Models discussed in this paper include the Agricultural Runoff Management II: Animal Waste Version (ARM II) model; the Utah State (UTAH) model; the MWASTE model; the COLI model; and the Hydrological Simulation Program – Fortran (HSPF) model. Each of these models simulates bacterial die-off using first-order exponential decay (Chick's Law) or modifications thereof. The MWASTE model modifies Chick's Law by adjusting the die-off rate constant based on temperature, manure application method, and soil pH. The COLI model combines the Modified Universal Soil Loss Equation (MUSLE), Chick's Law, a cell density factor, and a temperature adjustment factor to determine the amount of bacteria lost from land-applied waste. The HSPF model allows both accumulated and fresh manure to contribute to bacterial losses from land applied manure. Both the ARM II and UTAH models utilize a form of the Freundlich isotherm to determine bacterial soil retention, while the MWASTE model uses an alternate equation to simulate bacterial soil retention.

Management practices that assist or deter bacterial survival and/or transport that should be included in NPS bacteriological models include manure land-application timing and frequency, buffer strips, storage or treatment facilities (both solid and liquid waste), and incorporation. Because most of these practices are not implemented across an entire watershed or subwatershed, distributed models that allow spatial variability to the field or sub-field level enhance the ability to evaluate management practices on bacterial survival and/or transport. None of the reviewed models allow this level of spatial detail. However, the MWASTE model allows simulation of buffer strips and application frequency, while the COLI model accounts for management practices through the USLE practice factor. The COLI model also accounts for incorporation of manure by averaging the bacterial concentrations of the manure and soil, thereby reducing the amount of bacteria available for runoff.

Further investigations are necessary to 1) quantify bacterial growth in manure applied to field soils; 2) determine field soil die-off rate constants; 3) evaluate the validity of the soil retention calculation assumption that equilibrium is achieved between bacteria in soil and soil solution; 4) formulate quantitative methods that describe partitioning of bacteria during transport to model the effects of settling and resuspension of bacteria (both overland and in-stream), which could improve bacteriological modeling of management practices such as buffer strips, as well; 5) improve understanding of the effects of incorporation on bacterial survival and transport; and 6) incorporate detailed spatial variability into future bacteriological models to allow adequate representation and evaluation of management practices.

Keywords: bacteria, nonpoint source pollution, modeling, manure

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Introduction

Approximately 66% of impaired stream segments in Virginia are impaired due to fecal coliforms (VDEQ and VDCR, 1998). Nationally, an estimated 19,811 river miles, approximately 8% of the nation's impaired streams, are impaired by pathogens (USEPA, 1998). The role bacteria play in water impairment encourages model development to describe bacterial transport to and in water bodies, particularly in light of the Total Maximum Daily Load (TMDL) program. Development of accurate nonpoint source (NPS) bacteria models will improve estimations of bacterial loadings to waters and provide the ability to evaluate the effects of implementing best management practices (BMPs). These models should be user friendly and interactive allowing government agencies and consultants to assist farmers, watershed organizations, and planners in developing strategies to reduce bacterial pollution of surface waters. This paper provides background information regarding bacteria, processes important to modeling bacteria, existing NPS models with bacteriological components, and suggestions for future research.

Background

Pathogens are disease-causing organisms, such as viruses and some bacteria. Enumeration of pathogens is often time-consuming, technically intensive, and costly; therefore, pathogen presence is often estimated through the utilization of indicator organisms. The presence of indicator organisms in a sample means only that pathogenic organisms may be present. Water quality standards, which vary from state to state, are typically based upon the presence/absence or concentration of indicator organisms such as total coliforms (TC), fecal coliforms (FC), and fecal streptococcus (FS). Because FC and FS are found in the intestines of warm-blooded animals, their presence is indicative of fecal contamination. Sources of fecal contamination include land-applied manure and sludge, manure from grazing animals, wildlife feces, combined sewer overflows, and septic systems. Currently, Simmons et al. (1995) are developing and utilizing DNA fingerprinting of *Escherichia coli* to determine sources of fecal contamination.

Crane and Moore (1985) summarized findings from the literature citing nineteen environmental parameters that affect bacterial survival, such as solar radiation, temperature, soil moisture content, soil organic matter content, soil particle size distribution, waste application method, competition with other organisms, and nutrient deficiencies. The most important factors are generally considered to be temperature, soil moisture content, pH, solar radiation, available nutrients, and the waste application method (Crane and Moore, 1985; Reddy et al., 1981). An increase in temperature lowers the survival rate of bacteria (Reddy et al., 1980); a decrease in soil moisture content decreases survival rate of bacteria (Boyd et al., 1969); survival rate of bacteria is adversely affected if pH is outside of the 5.8-8.4 range (Lambert, 1974); and manure incorporation increases bacterial survival in comparison to surface application (Giddens et al., 1973). Edwards et al. (1997) found that in-stream bacterial concentrations were affected by seasonal influences and flow rate, with the highest concentrations occurring during the summer and highest flows. Additionally, Howell et al. (1996) and Sherer et al. (1992) concluded that fecal bacteria live longer in sediments than in overlying water and that bacterial survival is greater in fine sediments than coarse sediments.

Although research conducted to evaluate environmental factors on bacterial survival provide insight into bacterial behavior, the results of these studies must be considered prudently. Crane and Moore (1985) cautioned that researchers often do not or cannot measure all environmental factors that influence bacterial behavior during their experiments; therefore, measured bacterial response may be attributed to the wrong environmental factor.

Modeling

Processes that are important to bacterial survival should be included in a bacterial NPS model, including bacterial growth/die-off; sorption of bacteria to the soil matrix; partitioning of bacteria between water and sediment during transport; and the effects of management practices. In addition, if in-stream bacterial concentrations are of concern, then in-stream processes must be modeled because bacterial populations are dynamic and affected by growth/die-off and settling, as well as resuspension of bottom materials.

In the following sections, existing bacterial models are described in terms of modeling bacterial growth/die-off, partitioning between soil and soil solution, and effects of management practices. The models that are described include: the Agricultural Runoff Management II: Animal Waste Version (ARM II) model (Overcash et al., 1983a); the Utah State (UTAH) model (Springer et al., 1983); the MWASTE model (Moore et al., 1988); the COLI model (Walker et al., 1990); and the HSPF model (Bricknell et al., 1993).

Bacterial Growth/Die-off

Mancini (1978) and Crane and Moore (1985) described three commonly observed patterns of coliform die-off: first-order decay; bacterial growth followed by first-order die-off; and die-off rate that changes with time. The first-order decay equation used to describe bacterial die-off is expressed as Chick's Law (Moore et al., 1988):

$$\frac{N_t}{N_o} = 10^{-kt} \qquad [1]$$

where N_t = number of bacteria at time t; N_o = number of bacteria at time t_o; k = first-order die-off rate constant (1/day); and t = elapsed time since t_o (days). Modifications of Chick's Law by Mancini (1978), Polprasert et al. (1983), and Reddy et al. (1981) adjust the die-off rate constant for environmental impacts of temperature, solar radiation, pH, and/or soil moisture content. In their review of modeling enteric bacterial die-off, Crane and Moore (1985) stated that first-order decay has been used with "moderate success" to describe bacterial die-off.

The ARM II, UTAH, MWASTE, and HSPF models use Equation [1] to simulate bacterial die-off. The MWASTE model uses this equation both for stored and applied waste. Die-off rate constants for storage facilities are assumed to be 0.3/day, which is an average of storage die-off rate constants taken from literature (Moore et al., 1988). The adjusted MWASTE die-off rate constant for land-applied waste is determined as (Moore et al., 1989):

$$k = k1 * F_t * F_{ap} * F_{pH} \qquad [2]$$

where k1 = base die-off rate constant (0.5/day); F_t = temperature correction factor = $(1.0675^{(T-20)})$; T = temperature (°C); F_{ap} = method of application factor (0.50 for surface); and F_{pH} = soil pH factor.

Unlike COLI, which assumes that only fresh deposited waste contribute to bacterial runoff, the HSPF model accounts for accumulation of bacteria on soil and its contribution to runoff via the following equation (Bricknell et al., 1993):

$$SOQO = SQO(1 - e^{(SURO*WSFAC)}) \qquad [3]$$

where SOQO = washoff of bacteria from the land surface; SQO = storage of available bacteria on the surface; SURO = surface outflow of water; and WSFAC = susceptibility of bacteria to washoff.

In COLI, fecal coliform cell densities for manure and uncontaminated soil were determined from the literature. Total FC cell yield is computed using a combination of the Modified Universal Soil Loss Equation (MUSLE), Chick's Law, cell density factor, and temperature adjustment equation (Walker et al., 1990):

$$B_i = 11.8(10Q_d AF_i q_p)^{0.56} K_i LS_i C_i P_i D_i e^{-k_{20}\theta^{(T-20)t}} \qquad [4]$$

where B_i = number of bacteria cells eroded from area class i; Q_d = runoff depth (mm); A = watershed area (ha); F_i = fraction of total watershed area in the particular area class; K_i, LS_i, C_i, P_i = USLE factors for area class i; T = temperature of air or storage environment surrounding bacteria (°C); t = die-off period of bacteria cells (days); D_i = cell density factor; k_{20} = die-off rate constant at 20°C (1/day); and θ = regression constant (1.07 for several microbes). Four area classes are defined: surface applied manure areas, incorporated manure areas, pasture, and non-manured areas.

Partitioning Between Soil and Soil Solution

Gary and Adams (1985) and Sherer et al. (1988) showed that disturbing bottom sediments resuspends fecal bacteria in overlying waters. Despite these studies conducted on bottom sediments, the processes that dictate bacteria transport via water or sediment are not clearly understood. Coyne and Blevins (1995) determined that bacteria are transported as both free and sediment-sorbed organisms, and they also observed that bacteria preferentially adhered to fine materials, but no quantification or model was developed to describe the partitioning. Matson et al. (1978) determined retention coefficients (i.e. adsorption coefficients) for TC and FC in river sediments, but these coefficients are not necessarily applicable for field soils. VanDonsel et al. (1967) found that organisms

from runoff were in direct relation with counts in soils; the higher the soil bacterial count the higher the count of bacteria in runoff.

The ARM II and UTAH models utilize a simplified version of the Freundlich adsorption isotherm to determine the retention of bacteria in soils (Reddy et al., 1981):

$$MORT = K * MOSOL \qquad [5]$$

where MORT = organisms retained on soil; K = retention coefficient; and MOSOL = organisms present in soil solution. The retention coefficients were calculated based on the assumption of instantaneous equilibrium. Equation [5] describes the reduction of bacteria in runoff due to adhesion to the soil matrix. An alternate equation describing soil retention of bacteria is used in the MWASTE model (Moore et al., 1982):

$$Ft = FO(1-p)^{r} \qquad [6]$$

where Ft = bacteria remaining in soil; FO = number of original bacteria in soil; p = percent reduction factor of infiltration or runoff; and r = infiltration or runoff water depth. While Equations [5] and [6] describe the fraction of bacteria available for transport, neither of these equations describes the partitioning of bacteria in transport. The partitioning of bacteria during transport is important particularly to determine the effectiveness of BMPs such as buffer strips.

Effects of Management Practices

Management practices that assist or deter bacterial survival and/or transport include manure land-application timing and frequency, buffer strips, storage or treatment facilities (both solid and liquid waste), and incorporation. Timing and frequency of manure application can assist or deter bacterial survival and/or transport depending on climatic and soil conditions (Wheaton and Hale, 1980; Moore et al., 1988); buffer strips reduce the amount of bacteria delivered to surface waters (Young et al., 1980; Coyne and Blevins, 1995); and storage facilities allow time for bacterial die-off prior to land-application (Moore et al., 1988; Polprasert et al., 1983; Wheaton and Hale, 1980). Studies by Mazurak et al. (1975) and Khaleel et al. (1979) had conflicting results regarding the impact of manure incorporation on bacterial survival and transport.

Because bacterial survival and transport are influenced by a multitude of physical processes that have various quantifiable parameters, e.g., length of storage, bacteria are best modeled with theoretical models. Management practices, such as buffer strips, are often not applied to entire watersheds or subwatersheds. Detail of spatial variability may enhance or limit accurate representation of the effects of management practices on water quality, and thus, enhance or limit the evaluation of management alternatives for a watershed, farm, or field. Therefore, distributed models are best suited to account for field or sub-field scale variations in management practices affecting bacteria. Additionally, because bacterial growth and die-off can occur over extended periods (Simmons et al., 1995) and are vital aspects of modeling bacteria, a continuous model would be most appropriate to account for these long-term processes. Of the reviewed models, only MWASTE and COLI directly model management practices, meaning that explicit equations or variables exist for simulating the impact of a specified practice.

The MWASTE model utilizes an empirical equation developed by Moore et al. (1988) to simulate the effect of buffer strips on bacterial runoff concentrations:

$$PR = 11.77 + 4.26S \qquad [7]$$

where PR = percent removal of bacteria (not to exceed 75%) and S = buffer width (ft)/percent slope (buffer width > 10ft and 0 < buffer slope < 15%). Frequency of application is evaluated in this model indirectly in that frequency determines storage time (and therefore die-off in storage) and in the amount of bacteria on the land available for transport.

The COLI model addresses the issue of incorporation of waste by calculating an average of bacterial cells in the waste and in the soil, thus reducing the amount of available waste for runoff. Other management practices are included in this model through Equation [4] via the USLE practice factor, P.

Future Research

Observations indicate that bacterial growth rate can equal or exceed die-off rate days after initial introduction of manure to the environment (Howell et al., 1996; Crane and Moore, 1985; and Mancini, 1978), thus further investigations are necessary to quantify bacterial growth, particularly in field conditions. Field condition effects on die-off rate constants also need to be evaluated because most die-off rate constants have been determined for stream in bottom sediments.

Some bacteriological models account for bacterial adhesion to the soil matrix, but most of these models assume that equilibrium is achieved between the water and soil phases. This assumption is not necessarily true for field conditions; therefore, the validity of this equilibrium assumption should be investigated. Also, further study of the partitioning of bacteria during transport (both overland and in-stream) is necessary to formulate quantitative methods to describe the effects of settling and resuspension of bacteria.

Partitioning information could also improve bacteriological modeling of management practices such as buffer strips by quantifying the settled and/or resuspended materials within the strip during low and high overland flows. Further study into the effects of incorporation on bacterial survival and transport would also improve NPS model prediction of loadings to surface waters. As mentioned previously, many management practices are best modeled in distributed models, which allow relatively detailed spatial variability. Therefore, future bacteriological models should incorporate spatial variability to allow adequate representation and evaluation of management practices.

Data are needed to validate bacteriological components of NPS models once they are developed. Samples of runoff from manured fields with various management practices, such as incorporation or buffer strips, would allow researchers to compare model-predicted bacterial concentrations to actual observations. Taking random samples from manured fields over time would allow comparison of observed concentrations to predicted concentrations.

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